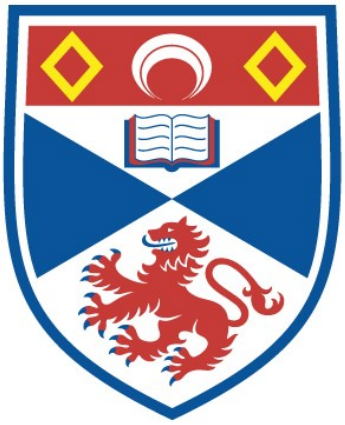
ASSESSING THE RELATIONSHIP BETWEEN POVERTY AND BIODIVERSITY, WITHIN THE CONTEXT OF LAND USE CHANGE IN THE SOLOMON ISLANDS

Tammy E. Davies

A Thesis Submitted for the Degree of PhD
at the
University of St Andrews



2014

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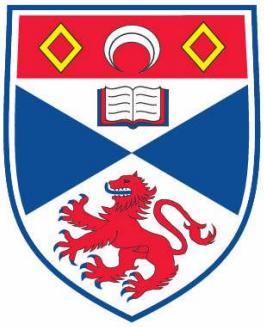

This thesis is submitted in fulfilment of the requirements for

the degree of Doctor of Philosophy

University of St Andrews

May 2014

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I, Tammy Davies hereby certify that this thesis, which is approximately 32,000 words in length, has been written by me, and that it is the record of work carried out by me, or principally by myself in collaboration with others as acknowledged, and that it has not been submitted in any previous application for a higher degree.

I was admitted as a research student in September 2010 and as a candidate for the degree of Doctor of Philosophy, Faculty of Science in September 2010; the higher study for which this is a record was carried out in the University of St Andrews between 2010 and 2014.

May 2014

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Tammy Davies Dr. Will Cresswell

Abstract

There is convergence at the international level that conserving biodiversity can contribute to poverty alleviation, but empirical evidence for this relationship is scarce. In this thesis I assess the relationship between poverty and biodiversity, within the context of land use change, using a case-study from the Solomon Islands. This interdisciplinary study is based on both social and ecological data, primarily collected through focus groups, household surveys and avian line transect surveys. Poor households in Kahua were characterised by fewer members of a working age and fewer male members. They were also found to own fewer assets, which were correlated to lower land tenure. Natural resources, including wild foods, were a crucial resource for the consumption and income for poor households, with evidence of wild foods buffering shortfalls in household consumption. The livelihoods of poor households were dependent on natural resources, whereas wealthier households relied on cash crops. The lower involvement of poor households in cash cropping suggests that the poor have less access to such income sources, possibly through a lack of initial land holding assets. Cash crop areas of monoculture cocoa were the most intensive land use in Kahua and were found to be a poor habitat for many bird species, including most endemics. Overall, the relationship between poverty and biodiversity was found to be complex, context dependent and influenced by various social and institutional factors. Household inequalities in access to land and resources indicate that a social-ecological trap may be occurring for poorer households in Kahua, possibly perpetuated by the livelihoods of wealthier households. More research is required in translating the concept of social-ecological traps into management actions, but this thesis concludes that this could be a useful concept for improving poverty alleviation and biodiversity conservation initiatives.

Acknowledgements

The last three and a half years have been both challenging and rewarding, and would not have been enjoyable, or even possible, without the support and expertise of those around me. I would first of all like to thanks my supervisors: Ioan Fazey, Nathalie Pettorelli, Will Cresswell and Guy Cowlishaw, who have been an excellent supervisory team and supported me from the beginning. Specifically I'd like to thank Ioan for introducing me to Kahua and your way of working; Nathalie for your constant support and welcoming me into the Institute; Will for always believing in me and saying the right thing at the right time; and Guy for your in-depth comments and thoughts. I wouldn't have made it without you, nor learnt as much and improved as a person without you all. Thank you for your belief in me.

The PhD would not have been possible without financial support from the Natural Environment Research Council and the University of St Andrews. I was also fortunate to receive additional support from Chester Zoo, Rufford Small Grants and Lubee Bat Conservancy, which enabled me to expand my fieldwork considerably.

In the field, my project would not have been possible without the hard work and dedication of my research assistants: James, Freda, Romano, Nellie, Mandy, Betty, Mevis, Eddie and Bassie, nor without the participation and cooperation of Toroa, Huni, Katoro, Nama, Borowe, Taratarena, Naopaworo and Paa communities. Particular thanks to Philip Marupa, Danny Wagatora and Andrew Wetara for your support and sage advice throughout the duration of my fieldwork. Not forgetting, my Kirakira family Lois and Noel, and all the staff at Freshwinds who made my weekends of data entry in Kirakira much more enjoyable. *Tangio tumas ufala everiwan*.

My time in the Solomon Islands was extremely tough and lonely and I have no doubt that I wouldn't have been able to complete it without the amazing support of Patti Duke and Val Stanley who were always there for me when I could make it back to Honiara. Your generosity and kindness knew no bounds and will never be forgotten.

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I owe heartfelt thanks to all my friends both near and far who have been unwavering in their support and encouragement, which has not only helped me keep my eyes on the prize, but also kept me sane throughout this process. Big thanks to Team Burns in St Andrews: Holly, Vikki, Marion and Alex, the PhD experience wouldn't have been nearly as much fun without you, and I can't believe we are almost disembarking from our journey down the river. Last but not least, my family who have given me the freedom to choose my own path in life, and always supported me in my decisions. I am truly fortunate to be able to follow my dreams, surrounded by such wonderful people.

Preface

This thesis stems from a personal interest in conservation, and more specifically improving natural resource management to be both socially and ecologically sound. My interest in this area was originally sparked while working for a marine conservation NGO in Madagascar, where conservation strategies proposed for the region were at odds with local livelihoods. More conservation work and a MSc later, I realised how much I actually enjoyed applied research, and the scope this provided to improve conservation and development efforts.

Thus, a personal aim of undertaking a PhD and writing this thesis was to develop and improve as an independent interdisciplinary researcher, to further my career in conservation. As a result, this PhD spans the natural and social sciences, and included the collection and analysis of both biological and social data, with chapters of this thesis written with publication in mind. While undertaking my PhD, I supervised a Masters student (Filip Ruzicka, UCL) who analysed the ultrasonic bat data I collected; I was involved in the study design, data collection, supervision of analysis and re-writing for publication, and have included the manuscript as Appendix A1.

Given my interest in applied research, I was keen to ensure my PhD addressed local concerns and needs in the Solomon Islands. Thus, I travelled to Kahua, my study site, prior to starting my PhD to develop appropriate research questions. I also received additional funding from Rufford Small Grants which enabled me to produce a small guide to the birds of Makira, which included an introduction to conservation and was distributed to all primary and secondary schools in Makira-Ulawa Province. Extracts of the bird guide are included in Appendix A2.

The Solomon Islands are by the far the most challenging place I have worked for a variety of reasons. The lack of infrastructure and rugged terrain made travel between study sites difficult. Torrential rain caused problems for data collection through disrupting surveys and preventing travel through rough seas and flooded rivers. This was further compounded by entrenched gender equalities and differing world views, which to some extent relate to the desires for money within communities and a general feeling that they were missing out on some sort of benefit. Allaying these feelings required intensive investments of time and resources throughout the data collection period. There is also a lack of background knowledge to the area, including an absence of basic guide books to local flora and fauna, so identifying anything other than a bird was complicated! Likewise, it was also these conditions and the understudied nature of the Solomon Islands that made Kahua an exciting site to base my study!

Naopaworo village, West Wainoni. Houses are made of traditional materials.

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Chapter 1: General Introduction

Introduction

The continued growth of human populations and per capita consumption have resulted in unsustainable exploitation of Earth's biological diversity (Rands *et al.* 2010). Broadly, biodiversity affects the properties of ecosystems and hence the benefits that humans obtain from them (Díaz *et al.* 2006). Thus the rapid rate of biodiversity loss, including disruption of ecosystems, loss of genes, species and functional traits, is of paramount concern for continued human wellbeing.

The tropics have experienced particularly dramatic species loss with persistently high rates of deforestation and environmental degradation associated with a change in land use from tropical forest to agricultural systems (Geist and Lambin 2002; Bradshaw *et al.* 2009; Flynn *et al.* 2009; Pereira *et al.* 2010). The rate of change in these regions is of particular concern, not only for the high levels of biodiversity these areas support (Myers *et al.* 2000), but also because they are populated by some of the world's poorest people (Wunder 2001). The persistence of extreme poverty and continued rapid loss of biodiversity appear intimately related. The use (and loss) of biodiversity, particularly through improved food production has benefitted many social groups (Millennium Ecosystem Assessment 2005). However, the consequences of biodiversity loss have been borne disproportionately by the poor (Cavendish 2000; Fisher 2004; Daw *et al.* 2011). This is because the poor tend to be directly dependent on biodiversity for their livelihoods, and have less access or capital to other livelihood opportunities, even when resources start to decline (Scherr *et al.* 2003; Dasgupta *et al.* 2005). Of particular consequence are declines in wild resources, biodiversity or ecosystem health that provide food, fuel, clothing, medicines and shelter, which have been linked to declines in rural health and welfare (Turner *et al.* 2013). Therefore, in terms of

contributions to livelihoods of the rural poor, it is not necessarily the diversity of natural resources, but rather the volume (in terms of extent and abundance) of a resource. However, in a more general sense, it is the diversity that is likely to be more important at the ecosystem level to enhance ecosystem resilience, with many species required to maintain multiple functions at multiple times and places in a changing world (Isbell *et al.* 2011). In this thesis, I refer to the precise elements of biodiversity that I am assessing, where appropriate, which is either natural resources or species richness using avifauna as a focal taxa.

At some levels the relationship between biodiversity and poverty is absolute: biodiversity underpins the delivery of essential ecological services on which the whole of humanity is dependent (Roe *et al.* 2014). Yet there is no linear relationship, and numerous studies have noted the dynamic and context-specific nature of the relationship (e.g. Kepe *et al.* 2004). However, there is an explicit assumption that conserving biodiversity (or reducing the rate of biodiversity loss) can help in efforts to tackle global poverty and enhance human well-being (Roe *et al.* 2014). The convergence of these joint biodiversity conservation and poverty alleviation goals are recognised at the international level through various policy frameworks, including the Millennium Development Goals and Convention on Biological Diversity Strategic Plan (2011-2020). At the local level, there is considerable divergence of opinion in the nature and scale of linkages between poverty and biodiversity, and how to address them. This is compounded by a lack of empirical evidence: there is surprisingly little observational or experimental detail describing poverty and biodiversity interactions in closely coupled human and natural systems in the rural tropics (Barrett *et al.* 2011), in addition to a lack of evidence for successful synergies between conservation and poverty alleviation initiatives (Leisher *et al.* 2013b). Ultimately, this weak evidence base is hindering both conservation and development goals. In terms of joint biodiversity

conservation and poverty alleviation efforts, priority needs to be given to protecting the elements of biodiversity, and the services it provides, that are of particular importance to the well-being of the poor (Roe *et al.* 2013). In addition, understanding who the poor are, the role biodiversity can play in alleviating poverty (e.g. through meeting basic needs, or as a safety net in times of need) is central to be able to prioritise conservation activities and ensuring the poor do not become further marginalised. In light of the considerable social and environmental change many Least Developed Countries are experiencing it is necessary to understand the relationship between poverty and biodiversity within the context of a social-ecological system.

Tropical islands present an excellent case-study for studying the relationship between poverty and biodiversity because they tend to have stronger linkages between ecosystem services and people (Millennium Ecosystem Assessment 2005). They also have reduced functional diversity (compared to continental areas) and are undergoing particularly rapid environmental change. For example, island deforestation rates have been recorded at almost three times the annual world average rate (0.8% compared with 0.3%) (FAO 1999). The Solomon Islands are a Small Island Developing State currently dealing with 'classic' challenges of sustaining rapidly growing populations, as well as more 'modern' problems of globalisation and global change (Reenberg *et al.* 2008). Crucially, as with many tropical island systems, the Solomon Islands remain desperately understudied, but are home to high levels of endemism making them an important focus for both conservation and development.

1.1 Thesis objectives

The over-arching aim of this thesis is to assess the relationship between poverty and biodiversity, within the context of land-use change, and how this relationship leads to a

social-ecological trap. I use Kahua, a remote area of the Solomon Islands as a case study to explore this relationship. To achieve this aim I will:

1) Clarify key terms, including poverty and biodiversity
2) Establish an appropriate measure of poverty, through evaluating whether a participatory approach can be used to gain locally-appropriate measures of household wealth inequality
3) Assess how household wealth relates to use of biodiversity through consumption and sources of income
4) Assess how biodiversity is changing with increasing intensity of land use
5) Use a broader systems view and overarching theory of social-ecological traps to explain the relationship between poverty and biodiversity and how it is changing within the context of land use change

1.2 Thesis structure

This thesis is interdisciplinary and has included the use of multi-methods to collect both social and ecological data. Therefore different methods have been used in each data chapter and as such there is no separate methods chapter, but rather the relevant methods are detailed in the appropriate chapter. Since data for this thesis are built around a case-study, I have detailed Kahua, my study site, in a separate chapter, including background information and specific details, to avoid repetition in subsequent chapters.

Following this General Introduction, in Chapter 2 I review biodiversity and poverty alleviation initiatives through synthesizing the latest literature from the conservation, development and interdisciplinary realms to assess reasons why there is a lack of evidence for success for joint initiatives. Through this review I clarify key terms,

including poverty and biodiversity. In addition I provide recommendations for poverty and conservation initiatives. Where possible, I incorporated these recommendations into the data collection and design of this thesis.

In Chapter 3 I outline my study site providing information on why Kahua is an ideal location for exploring the relationship between poverty and biodiversity in the context of land use change. I also provide relevant background information in order to provide a necessary understanding of the social and environmental context in which this study took place.

In Chapter 4, I gain an appropriate measure of poverty for Kahua. As highlighted in Chapter 2, definitions of key terms and multi-dimensional measures of poverty are important for effectively assessing the relationship between poverty and biodiversity. Therefore, in this Chapter I aim to establish a locally appropriate, multi-dimensional measure of poverty using a participatory approach. In this Chapter I determine: Can locally appropriate indicators of wealth be identified? Do these indicators represent variation in wealth within communities? Then to determine the validity of this approach I ask, are these indicators correlated with annual monetary income and expenditure? Finally I explore local inequalities through determining what the key predictors of poverty are.

In Chapter 5, I use the household measure of wealth established in Chapter 4 and relate it to the use of elements of biodiversity through consumption of wild foods and sources of income. In this chapter I assess whether household wealth influences consumption of wild foods, whether the consumption of wild foods differs between the lean-agricultural and non-lean season and whether household wealth influences income sources.

In Chapter 6, I then turn to the ecological system and assess how biodiversity changes with increasing intensity of land use, across five major land use types associated with local livelihood activities, using avifauna as my focal taxa. I address the following questions: Does avian species richness and the proportion of endemic species vary along a gradient of human disturbance? Does avian community composition vary along a gradient of human disturbance? Does avian diversity (species, microhabitat and feeding guild) vary along a gradient of human disturbance?

In Chapter 7, I summarise the aims, objectives, methodology, results and conclusions covered in previous chapters, and critically discuss the change occurring in Kahua in the broader context of a social-ecological system. I summarise key findings from this thesis and discuss the implications for conservation and development. I also indicate how the preceding chapters form a coherent body of work, the contributions I have made to this work, and finally how the work contributes significantly to the expansion of knowledge.

1.3 Thesis format

Numerous chapters in this thesis form the basis of publications (either in press or under review). These are acknowledged at the start of the relevant chapter, and are also listed here:

Davies T.E., Fazey I.R.A., Cresswell W., Pettorelli, N. 2013. Missing the trees for the wood: why we are failing to see success in pro-poor conservation. Animal Conservation 17 (4) 303-312

Davies T.E., Pettorelli, N., Cresswell W., Fazey, I.R.A. 2014. Who are the poor? Measuring wealth inequality to aid understanding of socioeconomic contexts for

conservation: a case-study from the Solomon Islands. Environmental Conservation DOI: 10.1017/S0376892914000058

Davies T.E., Fazey I.R.A., Pettorelli, N., Cresswell W., Cowlishaw, G. The importance of wild foods to rural households. PLOS ONE *Under Review*

Davies T.E., Clarke, R.H., Ewen, J.G., Fazey I.R.A., Pettorelli, N., Cresswell W. Impacts of land use change on endemic avifauna on Makira, Solomon Islands: endemics avoid monoculture. EMU *Under Review*

1.4 Ethics statement

The research was carried out in accordance with the ethics guidelines of the Geography and Sustainable Development Ethics Committee of St Andrews University who approved the methods and approach prior to data collection. A research permit was obtained from the Solomon Islands Ministry of Education, with support from the Kahua Association that represents communities in the study region. Prior to data collection, a meeting was held with all community members to carefully explain the purpose of the study, with ample time for respondents to ask questions. Due to the low level of literacy in the area it was not possible to obtain written consent, so informed consent was obtained verbally from all respondents who voluntary decided to participate in the study (Appendix A3). This procedure was approved by the Geography and Sustainable Development Ethics Committee of St Andrews University. Confidentiality was assured, and all responses were anonymised to reduce the risk of harm to informants.

Chapter 2: Literature review

Assessing reasons for the lack of evidence for effectiveness in joint poverty and conservation initiatives

Material from Chapter 2 formed the basis of the following publication:

Davies T.E., Fazey I.R.A., Cresswell W., Pettorelli, N. 2014. Missing the trees for the wood: why we are failing to see success in pro-poor conservation. Animal Conservation 17 (4) 303-312

Publication included in Appendix A5

A family outside their home in Nama village, East Wainoni. Woman on the right holds Sago palm fronds for re-roofing the house

2.0 Chapter Overview

In this chapter I synthesise the latest literature from the conservation, development and interdisciplinary realms to evaluate reasons for the lack of effectiveness for success in joint poverty alleviation and biodiversity conservation initiatives. Through this Chapter, I provide the broad conservation context for understanding the relationship between poverty and biodiversity, in addition to clarifying key terms that are essential to understanding the relationship between poverty and biodiversity. In this Chapter I bring together and discuss some of the major flaws highlighted in the literature that are hindering assessment of effectiveness of joint poverty alleviation and biodiversity conservation efforts. I then propose recommendations that address these flaws and that could help improve the evaluation of effectiveness in joint biodiversity conservation and poverty alleviation efforts.

2.1 Introduction

The current decline in biodiversity is of great concern, not only for the loss of its intrinsic value and the unknown impacts on ecosystem functioning, but also for the potential negative repercussions for human well-being (Cardinale *et al.* 2012; Reich *et al.* 2012). Addressing biodiversity loss and poverty are international societal and political goals (Sachs *et al.* 2009; Roe 2010). Each year the world spends around US$126 billion of official aid tackling global poverty and between US$8-12 billion on addressing biodiversity loss (Roe *et al.* 2011), yet in neither case are these resources considered sufficient to solve these challenges (Roe *et al.* 2011; Evans *et al.* 2012). The majority of the world's poor live in rural areas (IFAD 2010) where they depend disproportionately on biodiversity for their survival (Belcher 2013). This relationship has led to the explicit assumption that conserving biodiversity can help address global poverty, and in light of pressing challenges, such as population growth,

overconsumption and climate change, there is a strong need for further integration of the poverty alleviation and biodiversity conservation agendas (Sachs *et al.* 2009).

The integration of these agendas has so far proved more difficult and more expensive than many had hoped (Adams 2013). Conservation activities first started to address development issues in the 1970s, motivated by the substantial negative impacts on local people borne from earlier 'fortress' conservation activities (Adams *et al.* 2004). This 'people-friendly' approach, broadly termed Integrated Conservation and Development Projects (ICDPs) (Blomley *et al.* 2010), initially attracted substantial support from international development agencies and conservation NGOs, and was rather hurriedly seen as a panacea for conservation and sustainable development. However, early results proved disappointing and the approach rapidly fell out of favour (McShane *et al.* 2011; Miller *et al.* 2011). The ICDP label is now less common, but biodiversity conservation and poverty reduction continue to be addressed as dual goals; the extent to which largely depends on the perceived role of poverty in determining the status and threats to the intended conservation target (Sandbrook and Roe 2013). Adams *et al*. (2004) proposed a typology of four positions conservationists may take on the question of poverty: 1) poverty and conservation are separate policy realms; 2) poverty is a critical constraint on conservation, meaning it must be tackled to achieve conservation goals; 3) conservation activities must not compromise poverty reduction, and 4) poverty reduction is a goal, dependent on resource conservation. Recognising these fundamental differences in value positions can help contextualise the project's rationale, objectives and behaviour of different actors, particularly when faced with difficult trade-off decisions between conservation and development goals (Leader-Williams *et al.* 2011). Truly addressing the dual goals of biodiversity conservation and poverty reduction requires adopting the fourth position detailed by Adams and colleagues (2004), an approach known as "pro-poor conservation", and

defined as: a people-centred approach that has poverty reduction and livelihood security as core objectives and seeks robust conservation approaches to achieve these (adapted from: Roe *et al.* 2003; Roe and Elliott 2006; Kaimowitz and Sheil 2007). It builds on the poor's priorities and capabilities, effectively engages all stakeholders in addressing the underlying policy and institutional drivers of environmental degradation and empowers vulnerable groups with the assets, rights, and entitlements they need to improve their lives through sound environmental management (Hazlewood *et al.* 2004). Pro-poor conservation can take a number of different forms and encompass a variety of approaches, including: community-based conservation initiatives, direct payments (e.g. Payments for Ecosystem Services schemes, REDD+), and locally managed protected areas.

Despite biodiversity conservation and poverty reduction being addressed as linked challenges for a number of years, there is growing concern over the lack of empirical evidence for these endeavours (Barrett *et al.* 2011). An extensive review by Leisher *et al*. (2013b), although constrained by the limited number of studies with robust evidence of poverty impacts, was able to highlight ten conservation interventions that had contributed to poverty reduction. However, this review also found that only four of these ten initiatives benefitted the poorest (see Table 2.1 and Leisher *et al.* 2013b for further details). Despite the dearth of empirical evidence for success, there is still broad consensus among conservation professionals that there is a positive link between biodiversity conservation and poverty reduction (Roe *et al.* 2012). However, without empirical evidence, pro-poor conservation risks basing decisions on belief rather than evidence, repeating mistakes, and missing opportunities to replicate successes (Pullin *et al.* 2013).

Table 2.1: Summary of conservation interventions shown to provide poverty reduction benefits, and those which benefited the poorest (Adapted from Leisher et al. 2013b).

Intervention	Number of studies	Benefits the poorest?
Commercialisation of non-timber forest products	>50	Yes
Community forestry	>50	Yes
Payments for ecosystem services	10-50	
Nature based tourism	10-50	
Locally managed marine areas	10-50	Yes
Mangrove conservation	10-50	Yes
Protected area jobs	<10	
Agroforestry (integrating domesticated trees into agricultural landscapes)	10-50	
Grasslands (e.g. rotational grazing scheme)	<10	
Agro-biodiversity (initiatives that help farmers diversify the species & varieties of native crops on their farms)	<10	

In this Chapter, I synthesise the latest literature from the conservation, development and interdisciplinary realms. My aim is not to add to the debate on whether biodiversity conservation can contribute to poverty reduction, which distracts from the real task of improving the effectiveness of conservation-poverty reduction integration (see Roe *et al.* 2012), but rather to take the issue back to basics and evaluate reasons for the lack of evidence for effectiveness. Through this Chapter I provide the broad conservation context for understanding the relationship between poverty and biodiversity, in addition to clarifying key terms and concepts. I conclude this Chapter with recommendations to move pro-poor conservation forward.

2.2 Ambiguous use of definitions

Key concepts are often not explicitly defined in pro-poor conservation efforts (Lu 2010; Roe 2010; Roe *et al.* 2013), leading to vague objectives and preventing data aggregation and comparison (Kapos *et al.* 2008). Poor or narrow definitions are the root cause of limited empirical evidence in pro-poor conservation: how a concept is defined determines what is measured, and what is chosen to be measured determines how success is defined (Leisher *et al.* 2013a).

2.2.1 Poverty

Our understanding of poverty has evolved considerably from its original definitions, which focused on a lack of income or wealth (Sen 1993; Roe 2010). It is now viewed as a multi-dimensional concept, encompassing material deprivation, the lack of access to basic needs such as education, health, nutrition and food security, the absence of political autonomy and empowerment, as well as the lack of freedom of choice and social inequality (Chambers 1995; Shackleton *et al.* 2008; Sunderlin *et al.* 2008). Interest in multidimensional poverty measurement has been growing steadily over the last decade (Alkire and Foster 2011; Ravallion 2011; Ferreira and Lugo 2012) and mainstream poverty research has become more sophisticated in its handling of this concept, both through a diversification of methods, as well as by more inclusive processes of assessment that include the perspectives of the poor (Addison *et al.* 2009).

While the set of deprivations to be measured will vary depending on the social context, example indicators can include: school attendance, access to safe drinking water, ownership of durable assets and nutrition (Alkire and Foster 2011). Patterns of associations across multiple dimensions of poverty can be explored through simple tabulations, Venn diagrams or stochastic dominance analysis (Ferreira and Lugo

2012); they can also be aggregated into indices, such as the Multidimensional Poverty Index (Alkire and Foster 2011) or the Human Poverty Index (Dhillon and Kaur 2010). Hierarchical human needs (Maslow 1970) also presents a multidimensionality to poverty measurement, in addition to insights into what is required to improve wellbeing (Clarke *et al.* 2006). Poverty can be defined in either relative or absolute terms. Absolute terms measure poverty in relation to the amount of money necessary to meet basic needs, whereas relative poverty is defined in relation to the prevailing standards of living of other members of the society (Scheidel 2013). The decision whether to use absolute or relative poverty is ultimately a value judgement dependent on the main purpose for which the poverty measure is to be used. Where the relative definition is taken it is important to find a locally appropriate understanding of a certain social group or context (Scheidel 2013). In either case, the chosen dimensions across which poverty has been defined and measured must be explicitly stated, as conceptual and methodological differences in defining poverty can lead to the identification of different individuals and groups as poor (Lu 2010).

Although reports of pro-poor conservation studies discuss poverty as a multidimensional concept this is not translated into how poverty is measured, which remains confined to narrow income indicators (Leisher *et al.* 2013b; Scheidel 2013). This is despite income being known as an insufficient indicator of poverty reduction in isolation and having a low rank among the items by which the poor define their wellbeing (Chambers 1995; Brandolini *et al.* 2010; Ferreira and Lugo 2012). Furthermore, poverty reduction in the income-poverty sense, does not capture the complexity of livelihood and well-being-related objectives from local points of view (Berkes 2013). The Sustainable Livelihoods Approach (Scoones 1998) is one type of approach that can overcome such an issue, putting poor people at the centre of the analysis. Simplistic definitions of poverty have misdirected conservationists regarding

the question of community wants and needs, leading to narrowly conceived conservation benefits and incentives for local people (Murphree 2009; Berkes 2013).

The dispersion of a distribution of income, consumption or other welfare indicator, (or its inequality), is also an important dimension that is largely ignored. Inequality is moving to the forefront of the development policy agenda, in recognition of its relationship to poverty and poverty reduction (Melamed 2012): inequality directly determines the rate of poverty reduction, and certain types of inequalities (e.g. access to health care, education or markets, or civil and political rights) have a direct causal effect on poverty (Jones 2009). Equity has largely been addressed indirectly in pro-poor conservation through the implicit assumption about spreading the costs or benefits, or as a secondary concern, as with post hoc comparisons of the equity of outcomes (White *et al.* 2012). Pro-poor conservation would benefit from a focus on inequality as this would provide an incentive to focus on those groups which are left behind.

2.2.2 Biodiversity

Biodiversity is defined by the Convention on Biological Diversity as "the variability among living organisms from all sources including diversity within species, between species and of ecosystems" (CBD 1992). However, biodiversity is often used to refer to the amount of species and/or populations, or to specific elements of biodiversity rather than variety *per se* (Roe *et al.* 2013). Biodiversity, like poverty, is a fundamentally multidimensional concept (Lyashevska and Farnsworth 2012): it can be measured in terms of different components (genetic, population/species, and community/ecosystem; see Table 2.2). Each of these components has compositional, structural and functional attributes; these categories can be considered the 'three dimensions' of biodiversity (Lyashevska and Farnsworth 2012). This means that no single level of organisation

(e.g. gene, population, community) is fundamental, and different levels of resolution are appropriate for different questions (Noss 1990).

Table 2.2: Indicators of Attributes and Components of Biodiversity with a focus on measures that would be most useful in determining potential effects of human use on biodiversity (Agrawal and Redford 2006).

Attributes / Components	Composition	Structure	Function
Genetic	Allelic diversity	Heterozygosity Heritability	Gene flow Genetic drift Mutation rate Selection intensity
Population/ Species	Species abundance Biomass Density	Population structure, dispersion, and range	Fertility, Mortality, Survivorship, Life history Phenology
Community/ Ecosystem	Relative abundance of guilds or life forms, proportions of exotic or endemic species	Spatial geometry and arrangement of patch types	Disturbance regimes, Nutrient & energy flows, biomass productivity, patch dynamics

Due to the sheer number of species and the difficulty of sampling many ecosystems, measurements need to be simplified into tractable, quantifiable units that can be compared across time and space (Bradshaw and Brook 2010). Organism-based metrics that count the number of distinct species in a defined area (species richness) are the most common (Bradshaw and Brook 2010). However, species richness is arguably a narrow and poor estimate of biodiversity (Lyashevska and Farnsworth 2012). Metrics that accommodate a broader definition of biodiversity have been developed (summarised in Williams 2004), and there is a growing recognition of the

functional, phylogenetic and taxonomic aspects of biodiversity in conservation biology (Lopez-Osorio and Miranda-Esquivel 2010; Mouchet *et al.* 2010; Strecker *et al.* 2011).

There are two main problems with the use of the term “biodiversity” in pro-poor conservation. Firstly, where biodiversity has been defined, it is typically measured using a narrow perspective such as species richness, or a proxy that does not include any ecological information, such as perceptions of change in animal populations or attitudes towards conservation (Agrawal and Redford 2006). Broader approaches to defining and measuring biodiversity have yet to be incorporated into pro-poor conservation efforts. Conservation measures that aim to enhance a specific attribute or component of biodiversity may have unanticipated effects on other measures of biodiversity; multiple measures targeting specific combinations of attributes and components of biodiversity are therefore needed (Agrawal and Redford 2006). Secondly, the components used to frame biodiversity are often not clarified, rendering the collected data meaningless. Knowing the biodiversity (however measured) of one place, group or time is not useful in itself; it is the comparable measurements of biodiversity from multiple places, groups or times that can be used to answer crucial questions about how it might best be conserved (Purvis and Hector 2000).

2.3 Inappropriate monitoring

Another reason for the lack of evidence for evaluating effectiveness for pro-poor conservation is the lack of monitoring and inconsistent reporting of outputs or outcomes. Continual and independent evaluation of conservation interventions is a prerequisite to ensuring that conservation is appropriately targeted and effective (Saterson *et al.* 2004; Sutherland *et al.* 2004), in addition to ensuring that conservation fulfils its ethical responsibility to do no harm (Barrett *et al.* 2011). But as with mainstream conservation, pro-poor conservation initiatives have struggled with

designing appropriate methods for monitoring and evaluating project progress and outcomes (Blom *et al.* 2010). This lack of evaluation has restricted opportunities to learn and improve pro-poor conservation through adaptive management (Saterson *et al.* 2004).

Successful pro-poor conservation strategies occur across a variety of dimensions, including attitudinal, behavioural, economic and ecological (Wamukota *et al.* 2012). Data that cover only one or two of these dimensions have limited analytical value and can overlook trade-offs between outcomes (Daw *et al.* 2011; Brooks *et al.* 2012). For example, an extractive reserve may be considered a success by an economist based on increased income for local inhabitants, but a failure by an ecologist and an anthropologist based on critical population decline within the ecosystem and an absence of changed community values regarding conservation (Waylen *et al.* 2010; Brooks *et al.* 2012). Consequently, in order to effectively determine success of a pro-poor conservation strategy, measures are needed across the distinct dimensions, as defined by the expected outcomes (Wamukota *et al.* 2012). Furthermore, monitoring data for pro-poor conservation initiatives is rarely disaggregated for the poorest (or by ethnicity, gender, religion etc.), yet this would facilitate explicit assessment of how equity influences the ability to achieve project outcomes and produce a more nuanced picture of the intervention impacts on different groups (Halpern *et al.* 2013). Inappropriate monitoring can be attributed to four key problems: ambiguous definitions, donor pressures, lack of understanding between traditional scientific disciplines and lack of adequate reporting.

2.3.1 Ambiguous definitions

The use of ambiguous definitions cascades to vague objectives and difficulty in developing targets and indicators to gauge performance. Objectives of conservation

projects are often not clearly stated or linked directly to individual actions that might be monitored later (Bottrill *et al.* 2011). For example, community-level development activities, such as alternative livelihoods, are often emphasised as an indirect step toward effective long-term biodiversity conservation, but when the link between the activities and the aim are vague, projects tend to focus on ticking off activities, as opposed to monitoring the impacts of these activities (Sayer and Wells 2004). What, where and how to monitor, follows logically from clear and unambiguous objectives and questions (Lindenmayer *et al.* 2012).

2.3.2 Donor pressures

Effective monitoring is compromised through the influence of donors' demands and priorities (Bottrill *et al.* 2011) in three main ways. Firstly, there is a mismatch between the short time frame of funding and the long time frame needed to evaluate impacts on socio-economic conditions and the ecosystem, which often take longer than the funding period to undergo detectable change (Barrett *et al.* 2011; Pullin *et al.* 2013). Secondly, projects are influenced by donor priorities and emphasize particular kinds of objectives over others, with the objectives that are less attractive to funders left unstated or poorly articulated (Pullin *et al.* 2013). Pro-poor conservation projects have multiple objectives, and as a result the total monitoring can be a major drain on finite resources, with managers often reluctant to divert scarce resources from action to monitoring (Gardner 2010). In addition, the inarticulation of all objectives in the funding proposal means there are often inadequate funds available for monitoring the full range of outcomes (Kapos *et al.* 2008). Thirdly, neither donors (nor conservation organisations) have created a culture in which monitoring of outcomes is seen as desirable in its own right (Kapos *et al.* 2008). Both individual and institutional concerns about exposing shortcomings have served as a strong disincentive for critical evaluation. In some cases there is also an insidious disincentive for claiming or

demonstrating success in that perceived improvements may reduce the case for public, political and/or financial support (Pullin *et al.* 2013). However, Kapos *et al*. (2008) suggest that the capacity for rigorous analysis and synthesis should provide a strong incentive for evaluation to donors and practitioners alike.

2.3.3 Limited understanding between traditional scientific disciplines:

Monitoring is further compromised through a lack of understanding of social science research by natural scientists who often lead pro-poor conservation efforts (Lélé and Norgaard 2005). The collection of social data, such as people's behaviours or perceptions of change has followed natural scientists' affinity for quantitative data and large sample sizes that allow statistical analyses and broad generalisations (Drury *et al.* 2011). But poor understanding of the aims and scope of qualitative methods can lead to the false assumption that social data can be collected in a straightforward manner and interpreted at face value (e.g. that interviewees will always answer questions honestly), leading to misinterpretation of the results (Adams 2007; Homewood 2013). Natural science research approaches cannot simply be extended to the social science domain, as paradoxically an emphasis on quantification of social data may compromise data quality and validity (Adams 2007; Drury *et al.* 2011).

Ineffective application of social science research methods has also limited monitoring value due to a lack of meaningful participation. Given the range of different dimensions that need to be considered and in order to ensure data quality, the dimensions on which monitoring should focus should be defined by local people's priorities and local interest in contributing to and making use of the results (Homewood 2013). Local people's participation and inclusion in pro-poor conservation is rarely meaningful and in extreme cases can verge on coercive (Naughton-Treves 2012). Therefore what sets out to be a people-centred approach, in reality may only 'involve' local communities as

recipients of concessions and development assistance (Lélé *et al.* 2010). This ineffective application of social science research methods is ultimately detrimental to what pro-poor conservation sets out to achieve (McShane and Wells 2004). Successful initiatives require community objectives to be taken seriously and empowerment of all community members (Murphree 2009). Better application of social-science principles would help achieve this through developing an understanding of local aspirations, refraining from manipulating communities and thinking about trade-offs (Berkes 2013).

2.3.4 Lack of reporting

Many project implementers still do not report outputs or outcomes consistently, and consequently there have been few quantitative comparative evaluations (Waylen *et al.* 2010; Brooks *et al.* 2012). There is also a potential publication and reporting bias, with unsuccessful cases or metrics less likely to be published (Wamukota *et al.* 2012). This makes true comparative studies and analyses difficult, and strongly limits any attempt to describe relationships between poverty reduction and biodiversity conservation.

2.4 Recommendations

Evaluation of effectiveness of pro-poor conservation actions cannot be expected while these fundamental flaws persist, and I believe addressing these flaws must now become a priority. I propose five key recommendations to facilitate the collection of an evidence base that will enable the effectiveness of pro-poor conservation strategies to be determined (summarised in Table 2.3):

Table 2.3: Current flaws for evaluating pro-poor conservation and proposed solutions (see main text for details)

Problem	Solution
Ambiguous definitions	Unequivocal definitions of key terms (e.g. poverty and biodiversity) (Roe 2010)
Inappropriate monitoring	Rigorous monitoring that is efficient and appropriate for purpose (Kapos *et al.* 2008)
	Disaggregated data (Daw *et al.* 2011)
Donor pressures	Long-term approach for sustainability
	Fundamental shift in donor funding (long-term, flexibility etc.) (Bottrill *et al.* 2011)
Limited understanding between traditional scientific disciplines	Greater awareness and correct application of social-science principles (Drury *et al.* 2011)
Lack of reporting/communication	Systematic reporting of outcomes (whether successful or not) (Hirsch *et al.* 2011)
	Progress for the poorest groups reported separately (Melamed 2012)

1) Unequivocal definitions of key terms. Poverty and biodiversity need a multi-dimensional approach to their definition and measurement, and therefore success also needs to be measured across a variety of dimensions. This will lead to more nuanced questions, such as which groups of the differentiated poor depend on which elements of biodiversity (Roe 2010; Daw *et al.* 2011). Poverty, biodiversity and success are context-specific and the dimensions that have been used to classify them need to be clearly stated, alongside additional contextual conditions that can influence success, such as number of communities, size of population, level of dependence on biodiversity and local governance (Waylen *et al.* 2009; Wamukota *et al.* 2012). Baseline data also form part of the context and should be collected at the start of the project to provide a benchmark for comparison with data collected through monitoring of subsequent activities (Bottrill *et al.* 2011). Further work is required on developing and using standard multidimensional measures of biodiversity as the norm, rather than the exception. Species richness has been shown to be a poor surrogate for biodiversity

and consequently an urgent goal is to catalogue species with their significant functional traits in accessible databases to enable field-collected species lists to serve as a key to estimating biodiversity in its fuller meaning (Lyashevska and Farnsworth 2012).

2) Rigorous monitoring that is efficient and appropriate for purpose. This follows naturally from a thorough understanding of all the dimensions that need to be measured for poverty and biodiversity. A monitoring programme must be designed to be sensitive enough to detect incremental changes and capture the full range of potential outcomes across different dimensions (attitudes, behaviour, ecological and economic; see Margoluis *et al.* 2009). Appropriate monitoring data will reveal opportunities to learn and develop more responsible management practices (Gardner 2010). Appropriate monitoring also requires local inputs; equitable participation of all stakeholders is considered central to the adaptive management processes and ultimately leads to better results and sustainability (Jacobson *et al.* 2009).

Monitoring data should be both qualitative and quantitative, and collected on different levels and scales (e.g. individual, household, and community), as multilevel analyses are critical to understanding the dynamics at the different scales that can affect project outcomes (Brooks *et al.* 2012). Pro-poor conservation efforts should collect data in as disaggregated form as possible to facilitate analyses of inequalities and ensure the poorest and most vulnerable are not being left behind. Aggregated data can mask inequalities (UN Task Force 2012) whereas differentiated analyses (e.g. by gender, ethnicity, livelihoods and socioeconomic status, see Daw *et al.* 2011) can facilitate achieving conservation outcomes (Halpern *et al.* 2013) by elucidating the conditions where significant trade-offs are likely to occur (e.g. between the well-being of different people, either between or within communities, or between different outcomes, such as income and food security; Daw *et al.* 2011).

Since project outcomes may not be achieved over the short timescale of the project, indices based on outputs will always be needed (Jones 2012). Assessing the degree to which intermediate outcomes have been achieved can support adaptive management and provide insights on likely long-term effectiveness of interventions (Kapos *et al.* 2008; Pullin *et al.* 2013). For this to be effective, how the delivery of outputs is linked to outcomes needs careful consideration (see planning protocol in: Conservation Measures Partnership 2013). Jones (2012) suggests that for output measures to be more valuable for assessing project success, the linkages between outputs and outcomes, both in project proposals and reports, alongside the evidence upon which the assumption is based, should be explicitly stated (Jones 2012).

3) Long-term approach for sustainability. Pro-poor conservation efforts need to be based on long-term commitment, reflecting the time needed to make the project work (Blom *et al.* 2010). This will require a level of institutional change among organizations and agencies responsible for funding pro-poor conservation, through 1) providing more long-term and flexible funding; 2) incentivizing and promoting a culture of adaptive management and prospective ventures in trial and error (Bottrill *et al.* 2011), and; 3) being open to funding 'less attractive' projects that address the needs and threats to the poor and which are not built from pre-determined external viewpoints (Kaimowitz and Sheil 2007).

Poverty reduction can occur through improved governance and strengthening of local institutions and therefore for long-term sustainability, pro-poor conservation needs to emphasize community engagement, institution building and the devolution of authority and responsibility to local people (Belcher 2013). Possibilities and mechanisms for exit (or sustainability) strategy may only become clear after a number of years of successful

implementation, although it should be considered in the planning stage, with possible revisions based on achievements during the programme (Young 2008).

4) Greater awareness and correct application of social-science research methods. This could be achieved through collaborations across the social and natural sciences. This would improve pro-poor conservation through: 1) the adoption of meaningful participation (for further details see Burns *et al.* 2004)); 2) the facilitation of qualitative approaches that complement quantitative methods through acknowledging the limitations of different research methods in a social and cultural context (Drury *et al.* 2011), and; 3) improved understanding of local level processes and outcomes, particularly with regard to the complex formal and informal governance of common-pool resources. Use of common-pool resources is structured through institutions ("the sets of formal and informal rules and norms that shape interactions of humans with others and nature"; Agrawal and Gibson 1999); an analysis of these institutions could offer insights for developing appropriate pro-poor conservation activities, and could be achieved through the Institutional Analysis and Development framework (see Imperial and Yandle 2005; Ostrom 2007; Ostrom 2009). This would lead to the recognition of factors that can influence pro-poor conservation outcomes at the local level, including power relations and inherent biases of different actors in the research process (Drury *et al.* 2011).

Likewise, greater collaboration across the social and natural sciences would benefit poverty reduction development initiatives, which often result in biodiversity loss (e.g. oil palm versus old growth forests, dams and downstream impacts) and are generally based on unsustainable patterns of consumption and resource use (see Adams 2013). Greater collaboration between disciplines would facilitate appropriate evaluation of

biological and social outcomes, through mitigation of different languages and concepts between natural and social sciences (Ostrom 2009).

5) Systematic reporting of outcomes (whether successful or not). This will facilitate identification of effective interventions and the conditions under which they work more consistently, aiding replication and scaling-up of these successes in intelligent and evidence-based ways (Rands *et al.* 2010). Inevitably, there will be trade-offs in pro-poor conservation, but an open and integrative approach to acknowledging the trade-offs incurred by various choices and actions will provide insight and opportunity for genuine reflection, honest communication, and responsible action (Hirsch *et al.* 2011). Meaningful participation and multidimensional measurement of outcomes will help assess potential trade-offs, which should be assessed amongst all outcomes, and could lead to more resilient and sustainable conservation outcomes (McShane *et al.* 2011). To ensure the poor are benefitting from overall progress, data for the poorest groups should be measured and reported separately (Melamed 2012). While scientific publications and making data freely available are important for the wider scientific community, regular feedback of findings to stakeholders is also essential; it reaffirms that their involvement is being acted on in a transparent manner and also fosters social learning (Williams 2011).

2.5 Conclusion

In this Chapter, I have reviewed literature from across the natural, social and interdisciplinary domains to assess why there is a lack of evidence for evaluating the effectiveness in pro-poor conservation. This chapter demonstrates that effectiveness is being compromised in pro-poor conservation through a lack of attention to fundamental details in defining key terms and inappropriate monitoring. I propose recommendations that will improve pro-poor conservation through building upon clear definitions and

engaging in meaningful participation with rigorous monitoring and reporting of outcomes. Much has been learnt about the failure of ICDPs, but mistakes continue to be repeated, and I am certainly not the first to highlight the need for rigorous, systematic monitoring in conservation (e.g. Blom *et al.* 2010; Bottrill *et al.* 2011; Jones 2012). This highlights a disconnect between research and practice that urgently needs to be resolved leading to a culture of effective, rather than simply well-intentioned, conservation practice (Pullin *et al.* 2013).

Given that the challenges facing biodiversity and inequality worldwide show no signs of diminishing, the rationale for addressing biodiversity conservation and poverty reduction together is ever more important. In order to increase funding for conservation activities and to encourage donor confidence in conservation investments, there needs to be considerably more attention devoted to developing and applying robust and cost-effective approaches for evaluating success (Jones 2012). Conservation is currently marginal to the Millennium Development Goals, but building an empirical evidence base for pro-poor conservation could help influence the development of the post-2015 development goals and help development and conservation become more mutually supportive at both international and national levels.

In this Chapter I have brought together the latest literature from the conservation, development and interdisciplinary realms to identify reasons for the lack of evidence for effectiveness in joint biodiversity conservation and poverty alleviation initiatives. This Chapter has provided a broad introduction to the conservation angle of the relationship between poverty and biodiversity. In addition, I incorporate recommendations identified in this Chapter, where appropriate, into subsequent Chapters of this thesis. In the following Chapters I now move on to assessing the relationship between poverty and

biodiversity within the context of land use change, using a case study from the Solomon Islands.

Chapter 3: Study Site

Study site

3.0 Chapter Overview

In this Chapter I introduce my study site, Kahua, which is a region of Makira Island, in the Solomon Islands. This Chapter provides general background information on the Solomon Islands and Kahua, in order to provide a necessary understanding of the social and environmental context in which this study took place. This Chapter does not provide details of the methods used for data collection or analysis. Instead, and given the cross-disciplinary nature of the research, the different social and natural science methods are explained in the relevant chapters. In this Chapter I first provide justification of why Kahua was an ideal study site for this thesis. I then provide illustrated background information on the history and biogeography of the Solomon Islands, followed by a description of the Kahua study site, including its biological and demographic characteristics, to provide a comprehensive contextualisation of this thesis.

Crossing the Toroa river on way to Forest survey site

3.1 Rationale for selecting Kahua, Solomon Islands

In order to best explore the relationship between poverty and biodiversity within the context of land use change, I required a study site within a Least-Developed Country that contains high levels of biodiversity, has a population with a high dependence on the environment, and is undergoing rapid environmental change. The Kahua region of the Solomon Islands fitted all of these criteria. In addition, the components of biodiversity and the ecosystem services they provide are all the more critical in island systems (Millennium Ecosystem Assessment 2005), which makes understanding the relationship between poverty and biodiversity even more important from conservation and development perspectives. The Solomon Islands are, overall, an understudied region, presenting a unique opportunity to provide new data and insights about the biodiversity, ecology and underlying drivers of the threats to the biota on this island system. More specifically, the Kahua region of the Solomon Islands presented an ideal study site for various ecological, social and logistical factors. Ecologically, Kahua has particularly high levels of avian endemism, and relatively undisturbed coastal tropical rainforest due to a rare absence of commercial logging. Socially, Kahua has a degree of cultural distinction, with all communities speaking the same dialect (Kahuan; with different dialects spoken in adjacent areas on Makira Island). However, Kahua, and the Solomon Islands in general, present a challenging place to conduct research because not only are the Islands geographically remote and lack infrastructure, but also because the majority of land is customary owned. This means permissions must be sought and granted from local communities before accessing any land, and this process requires continual negotiation with multiple stakeholders.

Working in the Kahua region was facilitated through Ioan Fazey's prior work and strong relations with communities and a local grass-roots organisation, called the Kahua Association. The Kahua Association, which seeks to work across Kahua communities, is relatively unique in the Solomon Islands. Working with the Association helps facilitate research because it provides a bridge between communities and researchers, such as providing access to land and communities, and helping disseminate research findings. The Association also assisted by coordinating key project management tasks such as the selection of research assistants, study villages and research dissemination. This ensured greater control and responsibility of the research project by the local communities. The Kahua Association also acted as a buffer between researchers and communities and distanced researchers from the expectations and demands of local people, thereby greatly reducing the chance that the presence of external researchers would exacerbate aid dependency attitudes (Fazey *et al.* 2010).

3.2 The Solomon Islands

3.2.1 Background and History

The Solomon Islands is a sovereign country located north east of Australia and east of Papua New Guinea in the South West Pacific (Fig. 3.1). The country comprises a double-chained island archipelago over 900 islands, six of which constitute the majority of the land area, and stretch across approximately 1600 kilometres. The islands are located within 12 degrees latitude of the equator and more than 1500km from the nearest continent. They have a typical tropical climate, characterised by high and rather uniform temperature and humidity, with most areas experiencing abundant rainfall in all months (see Fig. 3.1; Solomon Islands Meterological Services 2013).

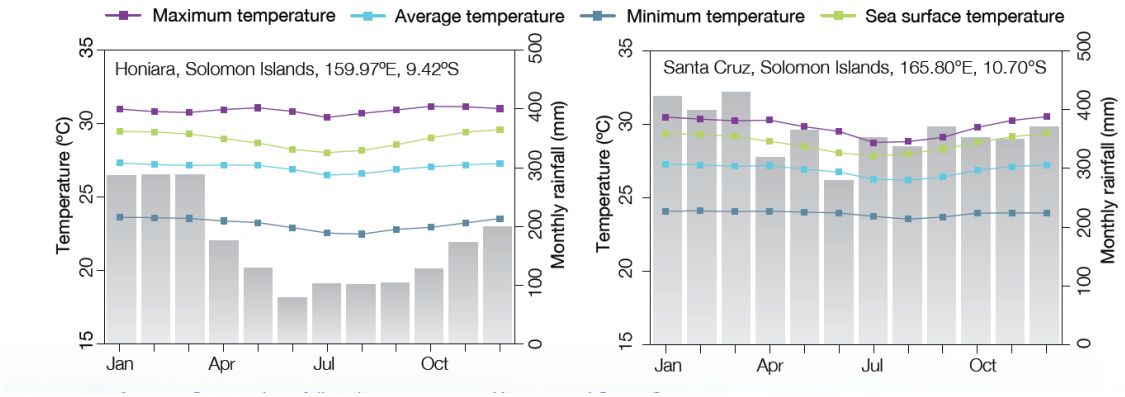

Figure 3.1: Seasonal rainfall and temperature at Honiara and Santa Cruz (PCCSP 2011).

The initial settlement of the Solomon Islands involved the people from Greater New Guinea by at least 29,000 B.P. (Matisoo-Smith and Robins 2004). Many modern-day descendants of these people speak what is often termed “Papuan” languages, which includes a remarkably diverse group of languages quite distinct from the more recently introduced Austronesian languages (Matisoo-Smith and Robins 2004). The Solomon Islands has one of the highest linguistic diversities in the world, with 75 spoken languages and a Linguistic Diversity Index of 0.965 (UNESCO 2009).

The first known European to visit the Solomon Islands was the Spanish explorer Álvaro de Mendaña in 1568. By the late 19th century, the islands were being exploited for labour to work in plantations in nearby areas of Fiji and Queensland, Australia. The Solomon Islands became a British protectorate in 1899. Under the protectorate, missionaries were safe to settle and started to convert people to Christianity. Today 92% of the population are Christian (US Department of State 2007). Prior to the protectorate being established head hunting and cannibalism were rife (Aswani 2000).

The Solomon Islands saw some of the fiercest fighting of the Second World War, which continued until the end of the war. The War established much of the limited infrastructure still in place today (e.g. roads, airport runways, bridges).

3.2.2 Biogeography

The Solomon Islands are part of the East Melanesian Islands biodiversity hotspot (Myers *et al.* 2000) and contain one of the last remaining tracts of coastal tropical rainforest (Bayliss-Smith *et al.* 2003). The Solomon Islands are a poorly studied region, with the majority of the information known about the fauna of this region collected during the Whitney South Seas Expedition (1921- c.1932). However, this unique biodiversity is under threat from rapid environmental change, with the Solomon Islands economy heavily based on extractive industries. Log exports comprise 46% of the economy, followed by minerals and fisheries (CBSI 2011). The logging industry is operating at unsustainable levels, and in 2011 almost seven times the sustainable harvest estimate of logs were exported, with just 4.5% from plantations (Hughes *et al.* 2010; CBSI 2012). Development in the Solomon Islands is hindered by this focus on a narrow productive base, in addition to a rapid population growth, low levels of investment, donor dependency, weak governance, and political instability (Hughes *et al.* 2010; Hayward-Jones 2014). The Solomon Islands is the poorest country in the Pacific region (OECD 2012), and is ranked 143 out of 187 countries in the United Nations 2012 Human Development Index. There are also significant physical constraints to the development in the Solomon Islands. The population of 500,000 is spread across more than 900 islands, consisting of hundreds of different identity groups with no national cohesion (Firth 2006). Within this context, traditional authorities continue to play a key role in the social, political and economic lives of rural people. In addition, inter-island transport and communication are poor, with the mountainous terrain of the larger islands leading to highly isolated communities. Most of the country

is continuously wet and in many places excessive rainfall is considered a constraint to agricultural production and infrastructure development (Macintyre and Foale 2004).

As with other areas in Melanesia, the Solomon Islands have also undergone rapid social change in relation to colonization, expanding trading opportunities, and the introduction of Christianity and western medicine. This has led to abandonment of traditional practices that controlled fertility (Dureau 1994), decreased violence-related mortality (Jackson 1978) and lowered rates of infant mortality (Lauer *et al.* 2013), resulting in a rapid population growth over the 20th century, that remains one of the highest population growth rates in the world (2.6% per annum; UNICEF 2011). The market economy was introduced to the Solomon Islands far later than in other developing countries (Furusawa and Ohtsuka 2006). Yet even in remote rural areas there is growing engagement with the cash economy, particularly through the payment of school fees, transport and imported foods, and on many islands cash crops have become important (Reenberg *et al.* 2008). The increasing effects of globalisation are leading to social and environmental change. The change occurring in the Solomon Islands is of particular concern from both conservation and development perspectives because the area is a biodiversity hotspot with high levels of endemism (Myers *et al.* 2000) and 84% of the population depend on subsistence agriculture (Mertz *et al.* 2012).

Traditionally the Solomon Islands have a male dominated society, and consequently there are pervasive gender inequalities and men continue to dominate all sectors of society from political posts to village chiefs (Fazey *et al.* 2011; Mataki 2011). Men also dominate most income generating activities and tend to have a lower commitment (than women) to spending on the health and education of their children (Gibson 2000; McMurray *et al.* 2008; Macintyre 2009). Cultural traditions remain strong, and as with other Pacific Island economies there is an emphasis on redistributive activities, with

most households involved in tribe-specific networks that give and reciprocate goods and services, termed the '*wantok system*' (Gibson 2006). Demands are rarely refused, and although the giver of the goods or services does not receive payment, the giver knows that it will be reciprocated when needed. While the wantok system can act as an important safety net, it also represents an unequal system that supports the interests of certain individuals (De Renzio, 2000) and is open to exploitation. For example, income earning individuals may find it difficult to save any money or to advance his/her economic position because of the heavy demands imposed by unemployed or "lazy" wantoks (De Renzio, 2000). Therefore the wantok system simultaneously creates a cultural resistance to sustained accumulation of material wealth (Foale and Manele, 2004). Please refer to Table 3.1 for further details on Melanesia specific terms.

Table 3.1. Definition of Melanesia specific terms

Melanesian term	Meaning
Wantok	Literally means "one talk" and is considered synonymous with "relative" (Foale 2008). It is an important concept associated with networks of distinct tribal, ethnic, linguistic, and geographic groupings in Melanesia (Nanau 2011).
Wantok system	Melanesian practice of demanding money or goods from relatives who are deemed to be capable of sharing (Foale 2008). Demands are rarely refused, and although the giver of the goods or services does not receive payment, the giver knows that it will be reciprocated when needed. In a country without a social security system, the wantok system provides for material care, a sense of identity and support during difficult times (Forster 2005).
Kastom	Generic term that refers to practices and traditions that are locality and wantok specific. Kastom is intimately connected with identify at all levels from individual to national (Tonkinson 1982)

3.3 Study site location

The research for this thesis was conducted in the Kahua region (Wards 12 and 13; $162^{0}0-162^{0}15E$, $10^{0}25-10^{0}40S$) of Makira Island (formally San Cristobal; Figure 3.2).

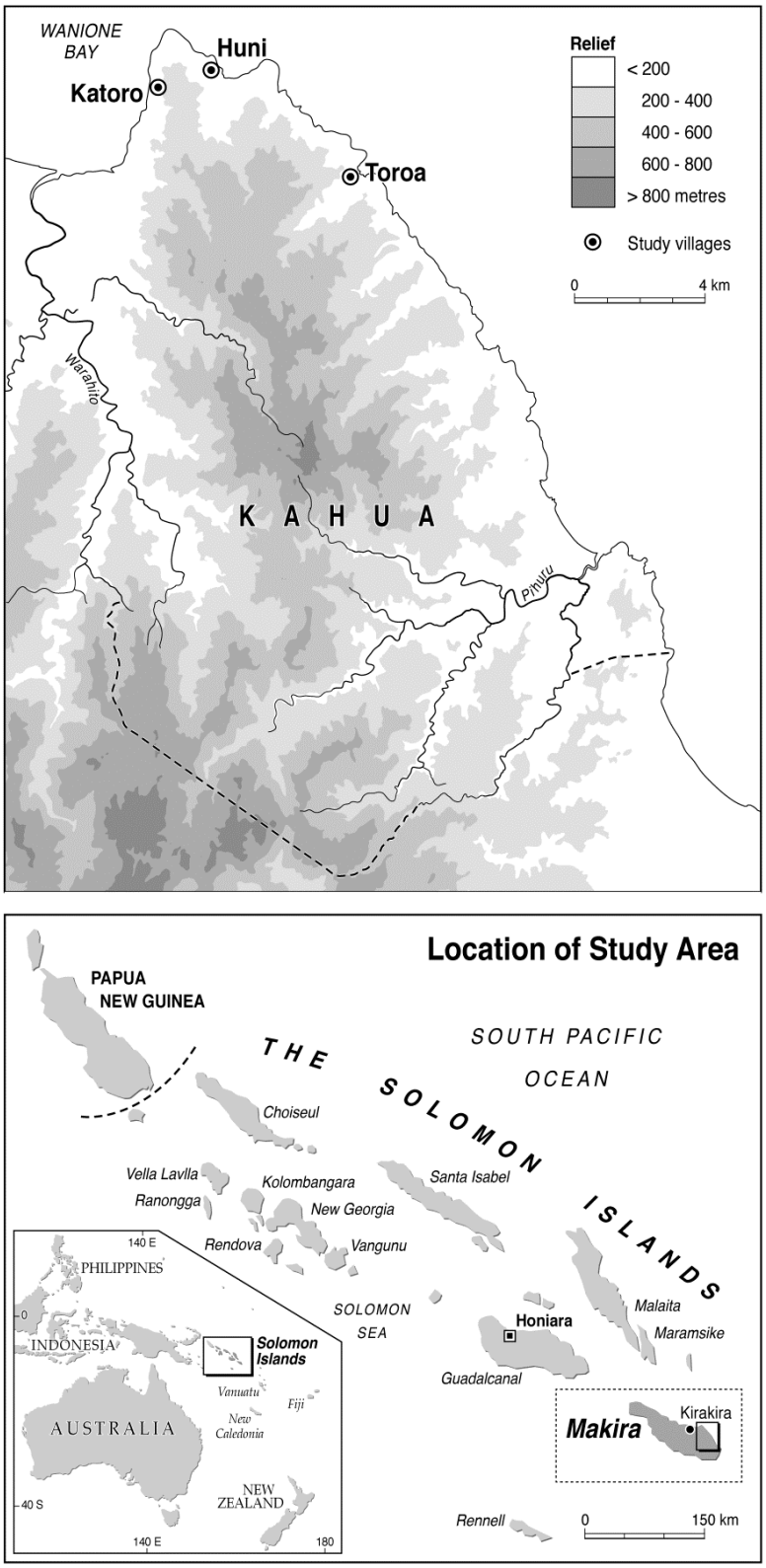

Figure 3.2: Location of Kahua study site

Makira is the fourth largest island of the archipelago with an area of 3191 km^2 and consists of a narrow coastal plain with steep forested central ridges with altitudes up to 1200 m (Allen *et al.* 2006). Makira has a wet tropical climate with little annual variation; average annual rainfall is 3.6–4 m, with up to 8 m reported in the higher elevations (Allen *et al.*, 2006). There is limited infrastructure, with no roads and only a limited number of high frequency radios for communication. Transport to the provincial capital of Kirakira (access to main market and hospital) is either by foot or fibreglass boats with small outboard motors (Figure 3.3).

Figure 3.3: Fibreglass boat used for transport around Makira Island. This picture shows a boat about to be launched from the wharf in Kirakira (the provincial capital of Makira)

Lack of market access is a major constraint to development in the region (Allen *et al.*, 2006). Kahua is experiencing rapid social and environmental change through an increasing population, desire for monetary prosperity, a loss of social cohesion (Fazey *et al.* 2011) and a loss of traditional methods of natural resource use and management (Ministry of Environment Conservation and Meteorology 2008).

Makira contains globally significant biodiversity and is an important area of endemism. It is part of a high priority Endemic Bird Area (Birdlife International, 2013) and contains 18 extant endemic bird taxa, which is more restricted-range bird species and subspecies than any other area of comparable size in the world (Danielsen *et al.* 2010). Kahua is one of the only regions on Makira not to have experienced commercial logging, but the region is still undergoing rapid environmental change. Changes in primary productivity have been detected suggesting significant ecological change at a landscape scale (Garonna *et al.*, 2009) and at the local level, the availability of forest and marine resources may be declining with reports of falling crop yields and increasing incidence of pests and diseases (Bourke *et al.* 2006; Fazey *et al.* 2011). One endemic species of bird (Thick-billed Ground Dove *Gallicolumba salamonis*) from Makira has been reported as extinct, and the Critically Endangered Makira Moorhen (*Gallinula silvestris)* has not been sighted since 1953 (Dutson 2011; BirdLife International 2013). Makira is particularly understudied region of the Solomon Islands, and there have been no recent surveys to establish population status or viability of many species, including the Critically Endangered Makira Moorhen.

The Kahua region has approximately 4500 inhabitants across 42 communities, mostly located on the coast (Figure 3.4).

Figure 3.4: Huni village viewed from the sea

The main livelihood strategy is subsistence agricultural production, supplemented by fishing and exploitation of a wide range of species. The main food crops are yams (*Dioscorea alata* and *D. esculenta*), taro (*Colocasia esculenta*), sweet potato (*Ipomoea batatas*), and banana (*Musa* spp.). The Pacific has a long history of swidden (shifting) cultivation (Mertz *et al.* 2012) which is employed on Makira with lands rotated between cultivation (garden) and fallow, with large fruit and nut trees preserved throughout the landscape. I use the Melanesian term 'garden' in this thesis to refer to land used for small-scale agriculture (Fig. 3.5). Gardens differ from the 'home-gardens' found in many areas of Asia, which generally refer to land cultivated around the family home, where typically only supplementary foods, such as fruits may be grown. Melanesian gardens are often not located close to houses or villages, and because subsistence agriculture is typically the main livelihood activity, they tend to be larger areas of land than home-gardens (Mertz *et al.* 2012).

Figure 3.5: New garden area cleared in the forest (Left); Recently planted garden with sweet potato crops (Right)

There are gender-differentiated roles associated with agriculture: women are the primary farmers (Pollard 2000) and also the main gatherers of wild plants, which are often collected on the way to or from the garden. Men are involved in some agricultural activities (e.g. clearing of new gardens) and are heavily involved with plantation (cash) crops. Hunting is a male activity and culturally important, with wild pigs required for village feasts and reaffirming cultural ties (Miles 1998). Young boys also engage in hunting birds and flying foxes. Only traditional hunting methods are used (e.g. spears and dogs for wild pig, sling-shots or bow and arrow for birds and bats) since guns were outlawed in the Solomon Islands in 2003 (Wainwright 2003). Fishing is usually conducted with hook and line, from a three metre dugout canoe with no outrigger (Fig 3.6). Men are the primary fishers, with women involved in the collection of shells. However, the Kahua people are traditional forest people, only moving to the coast during the colonial period (Scott 2000). There are no fringing reefs in Kahua, with the sea floor dropping off quickly and often leading to large swells and surf during bad weather. As such, fishing is not a major livelihood activity in Kahua, compared to other areas of the Solomon Islands (e.g. Aswani 2005; Brewer *et al.* 2009).

Figure 3.6: Man going fishing in typical dugout canoe with no out-rigger.

There can be large swells in Kahua making sea travel difficult.

Interaction with the cash economy in Kahua is limited, but increasing, mainly through the payment of school fees, transport and imported foods. Households engage in a range of income generating activities including the sale of agricultural produce (Fig 3.7), handicrafts and livestock (chickens and pigs), and the production of cash crops (copra and cocoa). Participation in these income-generating activities requires access to land, which is predominantly customary-owned across Melanesia, with tenure established through genealogy (Fazey *et al.* 2011).

Figure 3.7: Small markets are becoming increasingly common in Kahua. The items for sale in this photo are sweet potato, green coconuts, pineapple, and some cooked goods (in the plastic containers).

Kahua's geographic isolation and economic marginality means that its communities are dislocated from political structures and processes and the region receives very little assistance from external institutions (Fazey *et al.* 2010). Within this context, traditional authorities, such as village chiefs, continue to play a key role in the social, political and economic lives of Kahua people. The Kahua Association is therefore an important mechanism for cross-community governance. The Kahua Association aims to achieve a more united, locally based, and sustainable approach to development (Kahua Association 2005). While the Kahua Association is a fairly young organization, it has already successfully created more effective cross-community discussions and more equitable decision-making, such as when communities have been approached by logging and mining companies (Fazey *et al.* 2010).

Who are the poor? Appropriately measuring poverty

Material from Chapter 4 formed the basis of the following publication:

Davies T.E., Pettorelli, N., Cresswell W., Fazey I.R.A. 2014. Who are the poor? Measuring wealth inequality to aid understanding of socioeconomic contexts for conservation: a case-study from the Solomon Islands. Environmental Conservation

DOI: 10.1017/S0376892914000058

Publication included in Appendix A6

Female focus group in Borowe village, discussing indicators of poverty

4.0 Chapter Overview

In Chapter 2, I highlighted the importance of clearly defining and using a multidimensional measure of poverty. There was no background information on poverty in Kahua, and as with other remote rural areas, typical assessments of poverty would have been constrained by limited engagement with a cash economy, complex family and tribal ties, and an absence of basic infrastructure, including electricity and therefore white goods that are common on household asset lists. Thus, in this Chapter I evaluate the value of a participatory approach to identify appropriate indicators that can be used to measure poverty, in addition to providing insights into poverty in a manner that is locally-appropriate. The measure of poverty I use in this Chapter is relative across all households within communities in Kahua, and therefore to reflect this relative measure of poverty, I use the term 'wealth inequality' throughout this Chapter.

4.1 Introduction

Conservation interventions aimed at improving the sustainability of natural resource use take place within a complex and dynamic ecological, economic, and social landscape (Dawson *et al.* 2010; Rissman 2011). Understanding these complexities is important for the design of successful conservation interventions, especially in areas with high degrees of inequality, to ensure conservation interventions do not inadvertently further disadvantage vulnerable people (Lawlor *et al.* 2010). In response to the failure of 'fortress' conservation efforts that often had substantial negative impacts on local people, many conservation projects now aim to work with local communities (community-based conservation) and include social objectives, such as poverty reduction, as part of their aims (Hutton *et al.* 2005). However, too frequently community-based conservation initiatives are implemented without fully understanding the local socioeconomic context (Homewood 2013). This ignores the heterogeneity of stakeholders and important factors, such as gender, ethnicity, religion, livelihoods, and

reliance on biodiversity, that affect how people are able to respond to and interact with conservation initiatives (Agrawal and Redford 2006). Failing to recognise these differences risks unequal distribution of costs and benefits from the intervention, with powerful elites capturing the majority of benefits, and the poor becoming further marginalised (Iversen *et al.* 2006; Saito-Jenson *et al.* 2010). This not only violates the ethical responsibility of conservation to do no harm (Homewood 2013), but is also likely to generate conflict between practitioners and communities, undermine support for conservation and ultimately compromise the long-term success of the intervention (Sommerville *et al.* 2010). Understanding the local socioeconomic context can help mitigate the unequal distribution of costs and benefits from conservation by informing the design of appropriate conservation initiatives and associated monitoring strategies (Barrett *et al.* 2011; Homewood 2013).

Given the unequal, and often highly-skewed distribution of resources and access to their benefits in developing countries, it is evident that researchers must analyse conservation benefits to the poor separately from the rest (or whole) of society (Daw *et al.* 2011), which requires wealth inequality to be measured so the poor can be identified. Thus far, the majority of conservation-based studies looking at poverty have used income as a measure of household poverty (Cavendish 1999; Ambrose-Oji 2003; Fisher 2004; Yemiru *et al.* 2010), mainly because income information is often readily available (Perry 2002). While monetary approaches can be useful, they do not provide a multi-dimensional picture of poverty that is necessary to develop targeted conservation and development strategies. Poverty is understood to be a multi-dimensional concept, incorporating elements of political disempowerment, a lack of access to critical investments such as education, and economic exclusion, rather than just low levels of wealth (Sen 1993; Chambers 1995; Gönner *et al.* 2007; McGregor and Sumner 2010; Alkire and Foster 2011). In addition, income data have limitations in

both accuracy and measurement, particularly in the context of developing countries where community-conservation projects are based, due to temporal fluctuations in income, inaccuracy in recollection, and sensitivity of certain types of income (e.g. illegal extraction). Income may not provide the best indicator of wealth inequality, particularly for short-term studies (see Nielsen *et al.*, 2012) often required in community-conservation efforts. Income data also fail to reflect the full amount of resources available to a household, including productive assets (e.g. livestock) and financial assets (e.g. savings), which can be used as insurance against income shortfalls (Brandolini *et al.* 2010; Nielsen *et al.* 2012).

Broader definitions and consequently measurements of poverty, such as asset wealth, are widely used in development economics (Carter and May 2001). Filmer and Pritchett (2001) developed an approach to asset wealth measurement in the absence of expenditure data that used an aggregate index based on durable assets owned by households to rank households. Assets provide a better picture of long term wealth because they accumulate over time, last longer and contribute to the productive capacity of a household through its resource stock (Moser and Felton 2009). Asset based poverty classifications better predict future income and expenditure than income and consumption measures (Liverpool-Tasie and Winter-Nelson 2011) and are the most important determinant of households choice of livelihood strategy (Ellis and Freeman 2004; Babulo *et al.* 2009; Nielsen *et al.* 2012). In addition, development studies that have examined the empirical relationship between initial inequality and subsequent growth have found a stronger effect of land and human capital inequality, than of income inequality, suggesting that asset inequality matters more (Birdsall and Londoño 1997; Rodríguez-Pose and Tselios 2010). Asset measures of wealth inequality may thus better inform conservation strategies than income or consumption inequality.

Typically, an asset measures approach uses presence/absence data on ownership of assets that capture living standards (e.g. radio, television, telephone, bike, motorbike, refrigerator and car; Alkire and Santos 2010) and infrastructure and housing characteristics (e.g. source of water, sanitation facility; Vyas and Kumaranayake 2006), which may form an index of socioeconomic status (or Material Style of Wealth) (Cinner 2009). Measurement is often limited to assets that are in some way measureable, with more intangible assets (e.g. social capital, access, and power) often ignored. Intangible assets are difficult to quantify because they are linked to the context, and to other complementary assets through which the intangible asset is deployed (Kaplan and Norton 2001; Hulme and McKay 2005). The asset approach usually involves an external assessor determining the kinds of assets to be assessed (Rakodi 2002). This external approach can be less informative for conservation studies, particularly where standard asset lists (e.g. radio, TV, fridge and bicycle) are inappropriate (i.e. all households lack basic assets).

Assessments of poverty can either be participatory or non-participatory. Participatory approaches are reflexive, flexible and iterative, and therefore better able than external approaches to facilitate exploring local knowledge and perceptions and encourage learning and empowerment at local levels (Chambers 1992; Cornwall and Jewkes 1995). Participatory approaches to poverty assessments are becoming more widely used in the conservation and development arenas, including methods such as wealth ranking that involves categorising households or individuals (Chambers 1994; Laderchi *et al.* 2003). However, people's own assessment of their condition may be biased as a result of limited information and social conditioning (Laderchi *et al.* 2003), or exaggerated in hope of receiving tangible benefits (Krishna 2009). In addition, despite the measures being nominally participatory, the level of participation is usually only extended to a few key stakeholders (McGee and Brock 2001; Naughton-Treves 2012).

Across many remote rural areas there are constraints to the use of typical assessments of wealth inequality, such as limited engagement with a cash economy, strong social networks and complex family and tribal ties, and absence of basic infrastructure and development. To gather appropriate and valid data, an approach that goes beyond monetary, presence/absence of standard assets, and key informant approaches is required. With this Chapter I aim to present a simple participatory approach to measure wealth inequality that does not pre-define the indicators to be used, but enables local people to identity them. This ensures a flexible and inclusive method, providing a perspective on poverty that is sensitive to local contexts, while simultaneously remaining straightforward and replicable for remote, rural community-conservation projects. I demonstrate this approach using a remote and data-deficient region of the Solomon Islands where there was no prior information on poverty. My specific objectives were to: (1) identify locally appropriate indicators of wealth, (2) assess whether these indicators are able to represent variation in wealth within communities, (3) compare the indicators with annual monetary income and expenditure, and (4) determine the key predictors of poverty.

4.2 Methods

4.2.1 Data collection

As detailed in Chapter 3, this study was conducted in the Kahua region of Makira, Solomon Islands. My research approach was broadly exploratory and inductive, with a combination of focus groups, discussions and a widely scaled household survey. It aimed to facilitate exploration of local knowledge and perceptions using deliberative methods that in Kahua are more closely aligned to social deliberative ways in which people communicate (Fazey *et al.* 2010; Kenter *et al.* 2011). Data were mostly collected by five local villagers, trained as facilitators and closely supervised by myself; these local facilitators were essential for maintaining trust between researchers and

communities, engaging with local communities, and translating information. Data collection methods were simplified to account for the facilitators' low levels of education and limited ability to simultaneously translate and record information, while also designed to capitalise on their local cultural and social expertise and knowledge (Fazey *et al.* 2011).

Focus groups were conducted in six villages across the Kahua region, with one coastal and one inland village sampled from each of the eastern, central and western localities (Figure 4.1, and refer to Chapter 3 for more information on Kahua study site). All village inhabitants were invited to participate. A total of 12 focus groups were conducted, which included a total of 109 participants, with an average of 9 per group. Focus groups lasted on average c.3 hours; all were conducted in the local language (Kahuan), and separately for men and women to manage gender-based power relationships. Due to low levels of literacy, informed consent was sought verbally from all participants at the start of the focus group. After an ice-breaker exercise, participants were asked to identify different wealth groups within their community. All groups identified three different categories: poor, average and wealthy. In groups, participants were then asked to identify items or characteristics that changed across these categories (indicators of wealth). Groups were asked to choose an item close to hand, such as leaves, to represent their chosen indicators which were then brought together for discussion. How these indicators changed across the wealth categories was then discussed. The total list of indicators from all focus groups was presented and discussed at a workshop, with 30 participants from across Kahua, where in groups of three, participants were given five stones and asked to rank the indicators they considered the best. This led to a shortlist of five asset indicators.

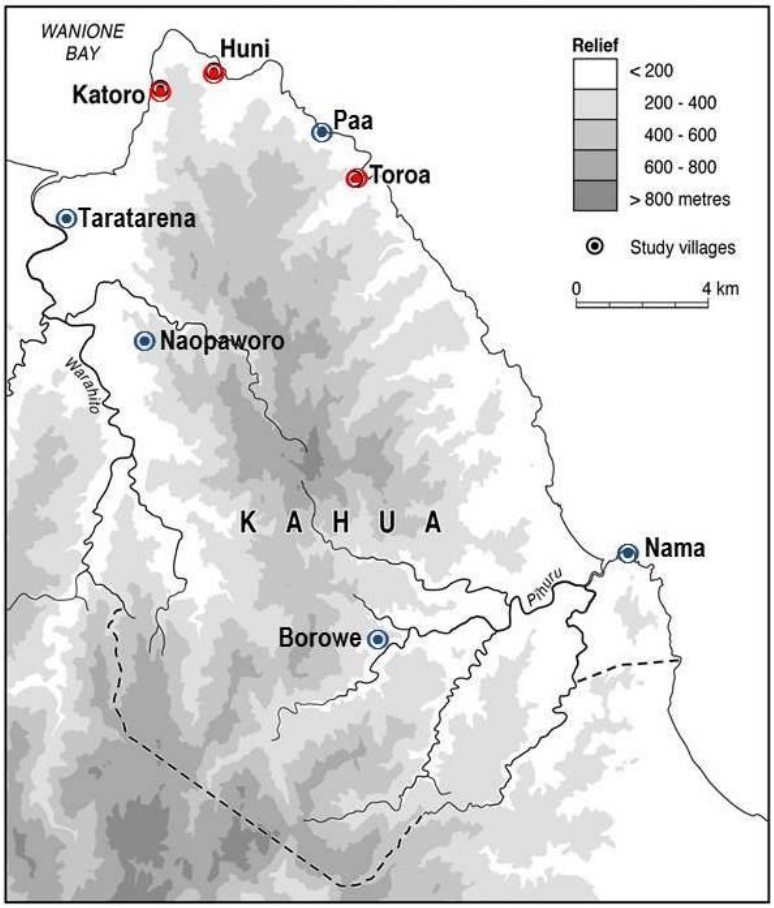

Figure 4.1: Location of study villages in Kahua. Katoro, Huni and Toroa communities (highlighted in red) were used for household surveys. Villages highlighted in blue were used for focus groups to establish indictors of poverty. Toroa village was used for both focus groups and household surveys.

A household survey was used to collect information on ownership of the top five asset indicators at the household level. A household was defined as people living together and sharing meals. The household survey was piloted in April 2011, refined and then conducted across 74 households from three communities in February 2012 and July 2012. The number of communities surveyed for the full data collection period was reduced to three communities due to the logistics and feasibility of walking between sites. All households were surveyed in each of the three communities (Katoro, Huni, Toroa). The head of the household was interviewed, or if unavailable another adult

from the household was interviewed. Basic information on household social structure including composition and levels of education was collected, and in January and July households were also asked to recall major sources of income and expenditure in the last 6 months. The latter data combined for a crude annual figure, focussed on major cash expenditures such as school fees and transport. Where there were differences in the information collected (e.g. household members, age) between January and July, the average value was used for analysis.

4.2.2 Data analysis

All data were analysed with R v2.15.1 (R Core Team 2013). Local indicators of wealth were identified as household ownership of number of pigs, chickens, coconut trees, cocoa trees and gardens. A Principal Component Analysis (PCA) was applied to household data on ownership of these locally defined indicators of wealth; the indicators are all continuous variables well suited to PCA. The factor scores from the first principal component (the vector that provides the most information about the variables) were used as the socioeconomic status index (wealth score) for each household. The higher the wealth score, the higher the implied wealth of the household. Differences in wealth score between villages were determined using an ANOVA.

To explore the variation in household demographics, a PCA was also applied to the household social structure data (number of household members, number of household members <18 years, age of household head, education of household head and dependency ratio (number of dependents [0-14 or >65 years] to the working-age population [15-64]) as a first step to determine the factors explaining most of the variation within the data. I did not consider the first axis of the household social structure PCA as a factor explaining the variation of the first axis of the household

asset PCA, as I aimed to assess how each component of the household asset dataset related to wealth inequality between households. To identify the main characteristics of the poor, the constructed household wealth score was then included as a continuous independent variable in a general linear model to explore the relationship between the wealth score and the household social structure: number of household members, age of household head, education of household head, gender of household head, proportion of males in the household and the dependency ratio. All possible combinations of main effects, followed by combinations of interactions were explored and then compared using Akaike's information criterion (AIC) values, which were compared among all possible combinations of explanatory variables. AIC is an evidence factor that is corrected for model complexity. Weighting AICs can be used to assess the best approximation to reality (model truth) by approximating Kullback-Leiber information loss to see how changing the model affects the fit (Bradshaw and Brook 2010), with a small value representing a better fit of the model to the data. To avoid model selection uncertainty where there were rival models, weighted averages of parameter estimates were calculated following Burnham & Anderson (2002). General linear models were used to compare the wealth score with income and expenditure, with the strength of the correlation assessed using Spearman rank correlation and R-squared values.

4.3 Results

Data were collected from 74 households across three communities (Table 4.1). Respondents had a mean age of 47.5 (±15.0) years, with an average of 5.6 (±2.4) years of education. Households had an average of 5 members (±2.1), with a mean of 2.4 (±1.6) children (those under 18 years).

Table 4.1: Population and social structure of the study villages in Kahua

	Villages		
	Toroa	Huni	Katoro
Households surveyed	32	27	15
Average people per household	5 ±1.5	6 ±2.6	5 ±2.1
Average years of education	6 ±1.4	6.2 ±1.9	4.7 ±1.9
Religion	South Seas Evangelical	Catholic	Catholic
Distance to Kirakira (km)	32.4	24.9	21.9
Sanitation	No	No	No
Water supply	Piped water (outdoors, shared)	No piped water	No piped water
Nearest clinic (km)	2.1	5.6	2.6
Nearest primary school (km)	0	0.7	3.9
Nearest secondary school (km)	14.5	7.3	4.3
Boat fare to Kirakira (US$)	$14	$8.5	$7

Focus group discussions indicated that wealthier households owned more of the locally defined indicators, which was corroborated with analysis of asset ownership (Table 4.2). PCA of these assets generated three components that together explained 71.4% of the variation (Table 4.3). The first component was composed of chickens with the greatest positive loading, followed by number of pigs, number of cocoa trees and number of coconut trees; these factors explained 36% of the variation in the data. The second component, with positive loading from number of coconut trees and strong negative weighting of garden number and garden size explained 20% of the variation, indicating less variation in gardens across the different wealth categories. The third component, explained 15% of the variation had a positive loading from number of gardens and number of cocoa trees and a high negative loading from garden size and number of coconut trees.

Table 4.2: Descriptive information gathered from all focus groups on how each indicator changes across the wealth categories (only main indicators identified at the workshop).

Indicator	Poor	Average	Wealthy
Pigs	No space for pig; lazy	Feed pigs coconuts; not always enough to feed them; no fence	More than 5 pigs; fence for pig; enough food to feed pig; sells for money
Chickens	No chickens; lazy	Some chickens	Lots of chickens; chicken coup
Coconut	No trees; asks for coconut; steals coconuts	Some trees	Lots of trees; always cooks with coconut milk
Cocoa	No trees	Some trees; 1-2 areas	Always sells to the ship
Gardens	Lazy; depend on others; don't plant much (cassava and banana)	2-3 gardens; plant 3+ crops; different crops in different gardens	5-10 gardens; doesn't use same garden each season; variety of foods

Table 4.3: Wealth factor scores from the principal components analysis of locally identified indicators of poverty

Variable	Wealth factors		
	1	2	3
Number of pigs	0.49	-0.07	-0.03
Number of chickens	0.53	0.13	-0.08
Number of coconut trees	0.45	0.37	-0.41
Number of cocoa trees	0.46	0.03	0.42
Number of gardens	0.24	-0.65	0.44
Average size of gardens	0.09	-0.65	-0.68

Based on the factor scores from the first principal component wealth scores for households ranged from -2.07 (poorest) to 5.40 (wealthiest) (mean = 0.00 ±1.5). Villages did not differ in wealth scores (ANOVA F=1.4, df=2, p=0.25) and therefore all analyses refer to grouped data.

A PCA of household social structure data generated three components that together explained 79 % of the variation (Table 4.4). The first component consisted of negative loading from number of household members, number under 18 years and the dependency ratio; the first component of this PCA explained 41% of the variation in the data. The second component had a positive loading from education of household head and negative loading from age of household head; the second component of this PCA explained 22% of the variation in the data. The third component had a strong negative loading from the proportion of males in the household; the third component of this PCA explained 16% of the variation in the data.

Table 4.4: Wealth factor scores from the principal components analysis of household social structure

Variable	Wealth factors		
	1	2	3
Household number	-0.56	-0.09	-0.05
Number under 18 years	-0.62	-0.02	0.06
Proportion of Males	-0.16	0.19	-0.94
Age of head of household	0.08	-0.67	-0.25
Education of household head	-0.14	0.68	0.08
Age dependency ratio	-0.51	-0.22	0.21

AIC model weights revealed the household social structure data, modelled as main effects, which best explained the variation in wealth scores were number of household members, age dependency ratio and proportion of males. A higher number of household members, lower age dependency ratio and higher proportion of males were associated with a higher wealth score. Based on Akaike weights, there was a rival model composed of number of household members and age dependency ratio. To avoid model selection uncertainty, weighted averages of parameter estimates were calculated (Table 4.5). There were positive correlations between wealth scores and monetary income ($p= 0.006$, $R^2=0.11$), the strongest being between wealth scores and monetary expenditure ($p<0.0001$, $R^2=0.24$, Fig. 4.2).

Table 4.5: Composite model of the strongest predictors of household wealth scores.

Parameter	Estimate	SE	90% CI Upper	Lower
Intercept	-1.72	0.39	-1.09	-2.36
Household number	0.37	0.05	0.46	0.29
Age dependency ratio	-0.004	0.001	-0.002	-0.01
Proportion of males	0.02	0.002	0.02	0.01

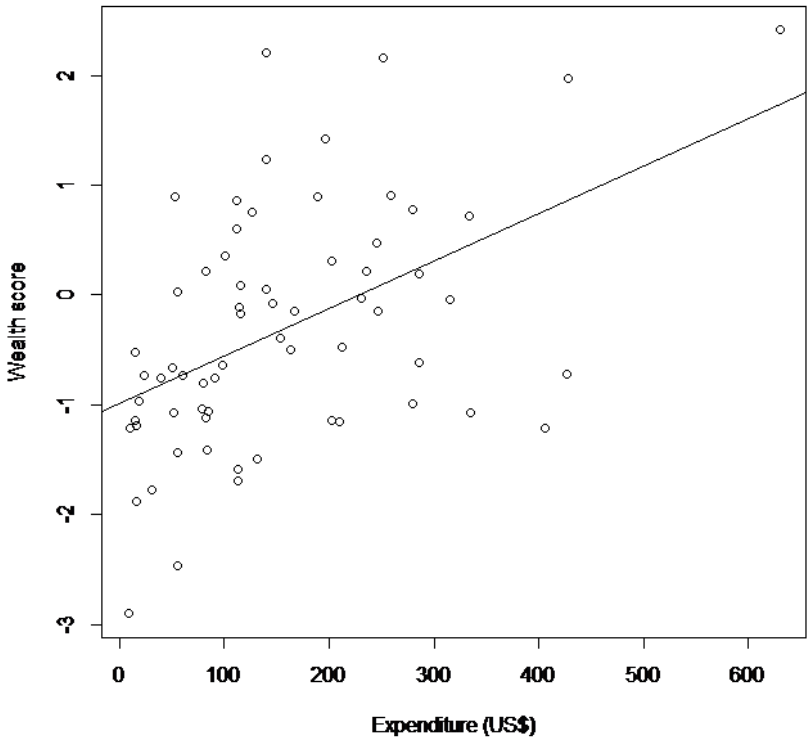

Figure 4.2: Plot of wealth score against annual monetary expenditure

4.4 Discussion

The participatory asset measurement method detailed in this Chapter avoided typical constraints to assessments of wealth inequality in remote rural areas, such as limited interaction with the cash economy, in addition to avoiding biases associated with external approaches. My approach provided key insights into characteristics of poor households where there was no prior information on poverty in a culturally sensitive manner that enabled participants to express their views on which indicators were important. Household asset wealth was particularly well correlated with household expenditure, which tends to be a better metric than income because households can smooth their expenditure during a temporary low-income period by borrowing or using savings (Perry 2002). As I only collected a crude measure of household expenditure, more detailed data would be expected to improve the strength of this correlation. However, the participatory approach to asset measurement I used goes beyond monetary metrics by providing better characterisations of the poor, which in turn provides further insights for the design and implementation of appropriate conservation projects and poverty reduction policies.

The poor in Kahua owned less of the locally defined indicators of wealth, particularly chickens, pigs, coconuts and cocoa trees. Little is known about rural poverty in Melanesia; however these assets reflect traditional Melanesian symbols of power. For example, pig ownership and pig killing traditionally conveyed status, wealth, and informal power in Melanesia (Miles 1997) and pigs are still culturally important in the region, remaining currency for major transactions (Glasse 1959; Miles 1998), including compensation payments and bride price that are commonly applied across the Solomon Islands. Food produce has long been used as a display of power, prestige and competition in Melanesia (Roscoe 2000), with the group having the largest number

and size of pigs, food crops and cooked food commanding the most respect (Nanau 2011).

Analysis of ownership of these assets also provided insights into the household characteristics of the poor, whose households had fewer members, a higher age dependency ratio and a lower proportion of males. In fact the poorest households contained no male members (i.e. older female living with young female child); other studies have also found female-headed households to be over-represented among the poor (e.g. Buvinić and Gupta 1997; Biewen 2006; Medeiros and Costa 2006). The participatory research approach I used enabled additional information to be elicited that would have been difficult to achieve otherwise. Focus group discussions revealed that people felt the poor's social position could be improved through hard work and a recurring theme was that the poor were lazy. For example, they might have access to land, but did not necessarily put in the effort to cultivate it, and therefore depended on exploiting the wantok system. Views that the poor are lazy are common (e.g. Lockwood 2002). However, although the poor may appear lazy, they may in fact be marginalised in some way, which means that they are unable to capitalise upon opportunities. For example they may have low personal empowerment (e.g. low confidence or social skills), or may not conform to social norms or abide by the same values as the rest of society (Applebaum 2003). In addition, the poor may not have access to land, for example if they are immigrants from other areas or families of men who have married into the region.

These results may help planning of appropriate community-based conservation and development initiatives to benefit the poorest. The locally defined indicators of wealth all require land, and therefore collectively can be considered to be correlated with land tenure. Thus, a higher wealth score can be seen to equate to ownership of, or access

to more land, and consequently conservation and development initiatives that are linked to land will naturally favour uptake by the wealthy, whereas the poor may be unable to invest or allocate land for such schemes (Corbera *et al.* 2007; Börner *et al.* 2010). Yet, current development activities in Kahua are focussed on the promotion of cash crops, activities which are unlikely to benefit poor households that have less land and are thus less likely to directly participate in these initiatives. In addition, monetization of resources can increase gender inequalities, adversely impacting women, which is of concern for poor alleviation efforts in Kahua where poor female-headed households could become further marginalised. The commodification of natural resources (e.g. through the introduction of cash crops) has shifted the Melanesian relationship with land from cultural to economic, and this shift is eroding social cohesion, with property rights currently a major source of conflict across Melanesia (Bonnemaison 1984; Foale and Manele 2004; Fazey *et al.* 2011). Given the assets and characteristics of poor households in Kahua, cash payments for conservation (e.g. payments for ecosystem services) are unlikely to be an appropriate conservation strategy there, because they could increase community conflicts, ultimately undermining conservation activities. Strategies that focus on small-scale resource management, balancing food security and conservation, such as agroforestry and locally managed marine areas are likely to be more appropriate for the social context in Kahua. Understanding the local socioeconomic context could help develop an appropriate enabling environment with interventions to improve people's capabilities and conditions, such as empowerment programmes and land reform (see McGregor and Sumner 2010).

Community conservation projects are often constrained by time and resources, with a limited portion of these available for monitoring activities (Gardner 2010). My asset-based approach within a participatory framework is well suited to community-based

conservation projects in areas with low levels of literacy and resources because it can collect valid and reliable data in an easily replicable manner. The participatory approach also provides an excellent starting point for discussing inequalities, and providing insights into how they can be alleviated or managed (Moser and Felton 2009). Findings from this approach can then be used to assist decision making on how best to target the poor and also as an input to other research problems, such as the relationship between wealth and observed behaviours, for example use of destructive fishing gear (Cinner 2009), uptake of conservation initiatives (Brandolini *et al.* 2010) and livelihoods (Reardon and Vosti 1995). For those community-based conservation projects that also aim to reduce poverty, longitudinal asset data can be used to monitor and determine transitions out of poverty. Although I used asset measures to provide an initial assessment of wealth inequality, this approach can also be employed in community-conservation projects before and after an intervention as part of monitoring activities to record longitudinal asset data or 'asset dynamics', which can help elucidate transitions out of poverty (Carter and Barrett 2006; Adjei *et al.* 2009). An approach for assessing household strategies for poverty alleviation has been pioneered by Krishna (Krishna and Shrader 1999; Krishna 2009).

Although my approach goes further than basic income measures of wealth inequality commonly used in community-conservation projects, it is still restricted to material dimensions of poverty. My approach was unable to distinguish between important capabilities, for example those who do not have access to land, and those who have access to land but choose not to cultivate it. Less tangible dimensions of poverty, such as social capital and power, were also not reflected in my assessment. My approach is not a panacea and further research is required into advancing poverty measurement that is better able to capture both tangible and intangible aspects of deprivations. However, it did provide insights into how poverty is viewed in the region, which appears

to be based heavily on traditional hierarchies and symbols of power (pigs), and therefore the locally identified wealth assets may also be a proxy for power; the extent to which these assets reflect power should be further explored. Social capital is the most commonly cited intangible asset (Moser and Felton 2009), yet kin and friendship networks are often the most important relationships that households mobilize to reduce vulnerability (Bacon 2005). The wantok system is an important informal institution in Melanesia for social cohesion and its contribution to balancing wealth inequality, and its ability to function as a support network, should be further explored using approaches that go beyond asset measures (Krishna and Shrader 1999).

This Chapter demonstrated an appropriate measure of household wealth and an understanding of poverty in Kahua. Poor households in Kahua owned less of the locally defined indicators of wealth, particularly chickens, pigs, coconuts and cocoa trees that were collectively correlated with limited land tenure. Poor households were also characterised by fewer members, fewer members of working age, and fewer male members than wealthier households. Having gained an understanding of poverty and established a measure of household wealth, I am now able to assess the relationship between poverty and biodiversity, through determining how household wealth affects the use of biodiversity.

The importance of wild foods to rural households

Material from Chapter 5 formed the basis of a manuscript which has been submitted as: *Davies T.E., Fazey I.R.A., Pettorelli, N., Cresswell W., Cowlishaw, G. The importance of wild foods to rural households. PlosOne Under review*

Young girl holding a Coconut crab (Birgus latro)

5.0 Chapter Overview

This Chapter builds on the measure of household wealth established in Chapter 4 to assess the importance of natural resources to households, on a tropical island where availability of natural resources is limited. Specifically, I assess households' direct use of natural resources, both in terms of consumption of wild foods and proportion of sources of cash income. Such direct uses of natural resources have been shown to be an important resource to the rural poor through providing opportunities for balancing nutrition and income generation, particularly in times of need (e.g. before the main agricultural harvest, during illness or crop failure).

5.1 Introduction

Tropical forests contain the highest levels of terrestrial biodiversity and also overlap geographically with large numbers of rural poor (Wunder 2001). Many forest products are common-property resources that require little capital or skills for their harvest (Angelsen and Wunder 2003), and consequently they are often an essential resource for the poor (Belcher *et al.* 2005). However, current rates of biodiversity loss are compromising the ability of ecosystems to sustain the current production of natural resources (Hooper *et al.* 2012), placing the poor at risk of further deprivation. The unacceptably high levels of both poverty and biodiversity loss in these areas has renewed interest in understanding the relationships between forests and poverty (Belcher 2013), with the extent to which forests could, and should, play a role in meeting the Millennium Development Goal's poverty alleviation targets gaining increasing attention (Hogarth *et al.* 2012).

Currently there is a lack of empirical evidence for the significant role of natural resources in the tropical rural household economy, which means provisioning ecosystem services that sustain livelihoods are routinely underestimated or ignored (de

Merode *et al.* 2004; Bharucha and Pretty 2010). Understanding these values are important as they can help shape policies that conserve and develop environmental assets for the poor in a targeted manner (Babulo *et al.* 2009). Consequently of particular interest is the use of natural resources that meet basic needs and sustain livelihoods, and in this respect the use of wild foods has received particular attention, primarily across Africa (e.g. Paumgarten 2005; Bennett *et al.* 2007; Shackleton *et al.* 2011). Wild foods, such as bushmeat, fish and plants, are often critical for providing opportunities for income generation and balancing nutrition (Harris and Mohammed 2003; Shackleton *et al.* 2011). The poor tend to be the most dependent on wild foods, both for consumption and income (Wunder 2001) and particularly in times of need (e.g. before the main agricultural harvest, during illness or crop failure) (Angelsen and Wunder 2003). However, wildlife is becoming increasingly depleted around the world (Fa *et al.* 2003; Milner-Gulland and Bennett 2003) bringing in to question the continued importance of natural resources for rural livelihoods, and subsequently the role natural resources are able to play in poverty alleviation. In parts of Asia where wildlife has been heavily depleted, people have switched to cash-earning jobs and alternative sources of food, such as domesticated protein (Bennett 2002) suggesting that natural resources lose importance in depleted environments. However, the poor are often excluded from, or lack access to alternatives (Cinner *et al.* 2009a), and are generally forced to choose lower-return livelihood strategies, even ones with continually declining returns (Barrett *et al.* 2006) and therefore, even when availability is reduced, natural resources may still remain an essential resource for the rural poor.

I use a case-study from the Solomon Islands to improve our understanding of the potential for wild foods to support poor households where availability of natural resources is limited. Tropical islands, such as the Solomon Islands, are inherently faunally depauperate compared to continental biotas (Wardle 2002; McConkey and

Drake 2006). Thus they offer a unique system to explore the importance of wild foods in the household economy where availability is constrained, providing insights for poverty alleviation, food security and conservation efforts. In this Chapter, I provide empirical evidence for the subsistence and income roles of wild foods, in relation to household wealth and seasonality. Poor households tend to be the most dependent on wild foods, and this relationship can be seasonal, particularly for tropical agricultural communities who tend to suffer from seasonal food shortages (e.g. before the main crop harvest) (Byron and Arnold 1999). During this lean period wild foods have been shown to be a critical resource for the poorest in helping them overcome shortfalls in consumption (Schulte-Herbrüggen *et al.* 2013) and income (de Merode *et al.* 2004). As such, understanding how wealth and seasonality affect the use of wild foods is central to developing conservation interventions and policy that support rural livelihoods. Based on studies from species-rich areas (e.g. Ambrose-Oji 2003; Paumgarten and Shackleton 2011) I predict that poor households will be most dependent on wild foods for consumption, particularly during the lean-season. Based on accumulated knowledge so far (e.g. Bennett 2002; Schulte-Herbrüggen *et al.* 2013), I also predict that wealthier households will have diversified their income sources, and rely less on exploitation of natural resources.

5.2 Methods

5.2.1 Study area

As detailed in Chapter 3, this study was conducted in the Kahua region of Makira, Solomon Islands. As with many tropical islands, the Solomon Islands are inherently depauperate in large-bodied vertebrate; the largest mammal present is the introduced wild pig with a mean weight of 35kg (*Sus scrofa*; weight from: Pangau-Adam *et al.* 2012), followed by the northern cuscus (*Phalanger orientalis*, mean weight 2.5kg:

Pangau-Adam *et al.* 2012) and numerous fruit bats (e.g. *Pteropus tonganus*, mean weight 0.5kg; Miller and Wilson 1997).

5.2.2 Data collection

Data were collected between January and July 2012, following a detailed pilot study (March-May 2011). The pilot study included participatory focus groups to compile a seasonal calendar for crops and hunger, which was then used to ensure the main study encompassed an equal number of lean-agricultural season (February-April) and non-lean season months (May-July). A total of 74 households were surveyed from three villages (Toroa n=32; Huni n=27; Katoro n=15, see Appendix A4 for village characteristics and social structure). Five local research assistants were trained extensively in social research methods prior to data collection, and assisted with data collection throughout the research period. An initial village census was completed prior to the main data collection to determine the number of households present in each village, and a preliminary indication of household demography. Based on these census data, every household received a unique household ID. Households were defined as people living together and sharing meals, and all households present and willing to participate were included.

5.2.2.1 Measuring use of wild foods and other natural resources

Data on both daily household dietary intake and household budgets were collected through interviews using the 24-hour recall technique (Bingham 1987), with households visited once every two weeks. Questions were addressed to the household member who prepared the food, or the household head. If neither of these respondents were present, the researcher returned later in the day or the following day. All foods and beverages consumed during the previous 24 hours, together with their mode of procurement (foraged/hunted, purchased or received as a gift), and any items

purchased, sold or given away by the household were also recorded. Respondents were also asked to recall any wild foods they had procured during the last 2-weeks. All foods were recorded in local names (Kahuan) with quantities recorded in local market measures with their associated market value, and subsequently converted to their economic value (US$).

A comparison of the value of wild foods harvested from the two recall periods, revealed two-week recall data to be 44% lower than expected from the 24hr-recall data. The 24hr-recall data were considered the more accurate, since the reported harvesting events occurred closer in time to the interview and there is a higher probability of interviewees remembering the harvest correctly (Beaman *et al.* 2005). Thus, I used the 24hr-recall data for consumption analysis, and only included the two-week recall data for compiling species lists.

Household composition was recorded during each survey. To control for variation in household size and composition, households were standardized to adult male equivalents (which takes into account that individual food needs vary by age and sex) using standard tables for moderate activity (as published in Smith and Subandoro 2007). Each measure of the household economy is expressed in daily US$ value per adult male equivalent (hereafter standardized US$). No labour costs were deducted as labour markets are absent making inputting the opportunity cost of labour time difficult (Babulo *et al.* 2009).

The 24hr-recall data were used to construct measures of the household economy, including household consumption (defined as the market value equivalent of all foods consumed), and production (defined as the market value equivalent of agricultural products and wild foods produced by the household, plus any income from sales,

remittances or gifts). Records of cash income, household expenditures and gifts were recorded separately. I explored the relative values of wild foods in the household economy in terms of household consumption and proportion of household income. Foods consumed were classed as wild, agricultural or imported. Wild foods were defined as those produced from an indigenous or wild biological resource, in some instances they may have limited cultivation or be sourced from non-natural systems (as for some edible leafy plants), but cannot be regarded as a conventional agricultural crop (Shackleton and Gumbo 2010). The most common agricultural foods were sweet potato, cassava, taro, banana and yam. Imported foods were defined as those foods not typically grown locally, and usually paid for at a store. The most commonly consumed imported foods were rice, tinned tuna and instant noodles. In addition to the consumed foods, we also recognised cash crops, which are defined in the same way as agricultural foods but are not consumed, such as cocoa and copra.

5.2.3 Data analysis

5.2.3.1 Measuring household wealth

Households were assigned a wealth score and a wealth category of 'poor', 'average' or 'wealthy' following the household measure of wealth, which was established in Chapter 4 (see Chapter 4, section 4.2.1 for full method details). Analyses were then conducted using both the continuous wealth scores and also between wealth categories.

5.2.3.2 Testing effects of household wealth and season

All data were analysed with R v3.0.1 (R Core Team 2013). I analysed differences in the household economy between the three household wealth categories (poor, average, wealthy) using ANOVA with post-hoc Tukey HSD tests, or Chi-squared tests as appropriate. To assess wild food consumption in relation to wealth and season I used linear mixed models (LMM) and generalised linear mixed models (GLMM) because

these can accommodate situations where observations are spatially or temporally non-independent (e.g. households within villages, and repeated observations through time) (Goldstein 1995). I assessed two measures of wild food consumption in relation to wealth and season: the probability of consumption (using GLMMs with a binary response variable of whether wild foods were consumed or not, a binomial error distribution and the corresponding logit-link function), and, if wild foods were consumed, their quantity (standardized US$, using LMMs with a Gaussian error structure and a log-link function). All models included village and the unique household ID as random effects. The wealth score of each household (continuous variable) was included as a covariate. Season (two factor levels: lean and non-lean) was included as a fixed effect and as an interaction term between wealth to account for any influence of season. All models were built based on my hypotheses and were compared on the basis of maximising predictive power for the minimum number of parameters using Akaike information criterion (AIC) (see Bradshaw & Brook 2010). Relative support for alternative models was evaluated with reference to the model with the lowest AIC value through both AIC difference ($\Delta AIC_i = AIC_I - AIC_{min}$) and Akaike weights for each model.

5.3 Results

5.3.1 Overview of the household economy

A total of 776 surveys were conducted with an average of 10.8 surveys per household (min 4, max 12, SD 1.39). Mean values for measures of the household economy across the full sample of households are given in Table 5.1. The figures for production (standardised US$2.70) and consumption (standardised US$2.02) indicate that, on average, all households in this study were above the extreme poverty line of US$1.25 per capita per day (as defined by the Millennium Development Goals), although, 67% of households fell below this line on at least one occasion. Gifts received by the household contributed to 3.3% of household production, with poor households

receiving more gifts than other wealth categories, although this difference was not statistically significant for wealth (ANOVA F=0.6, df=2, p=0.54) or season (ANOVA F=2.9, df=1, p=0.09). Cash income contributed 22% to household production, although income was sporadic (range 0-80.8, median=0), with income records for just 21% of all household surveys, so variation in the dataset was high (Table 5.1).

Table 5.1: Measure of household economy (standardised US$), across all households for each wealth category

	All households (n=72)	Poor (n=25)	Average (n=32)	Wealthy (n=15)
Consumption	2.02 ±0.97 (0.39-8.85)	2.18 ±1.1	1.95 ±0.95	1.88 ±0.82
Production	2.70 ±4.56 (0.39-82.6)	2.63 ±1.9	2.65 ±4.53	2.93 ±7.24
Income	0.58 ±4.40 (0-80.8)	0.33 ±1.14	0.61 ±4.40	0.99 ±7.22
Expenditure	0.46 ±1.08 (0-19.0)	0.39 ±0.79	0.40 ±0.82	0.75 ±1.78
Gift	0.09 ±0.45 (0-8.2)	0.11 ±0.60	0.09 ±0.33	0.07 ±0.34

*Means, standard deviation (SD) and ranges (for all households only) are given. Sample size (n) is listed in parentheses.

5.3.2 Wild food consumption

I found all households produced wild foods during the six month period. The majority of all produced wild foods were consumed by the same household (88%), thus I focus on consumption patterns. Terrestrial plants were the most frequently consumed comprising 86.6% of wild food records, followed by marine resources (11.7%), terrestrial vertebrates (1.5%), and riverine invertebrates (0.2%). Plants generally had a lower economic value (average standardized US$0.8) than marine resources (average standardized US$3.0) or terrestrial vertebrates (average standardized US$2.2), with

terrestrial plants accounting for 67% of the total wild food value, followed by marine resources 28.7%, terrestrial animals 4.1% and riverine invertebrates 0.1%. The top five terrestrial plant and animal species consumed are listed in Table 5.2 (see Appendix A8 for full species list). Wild foods comprised a relatively small proportion of household consumption (5%), compared to agricultural produce (68%) and imported foods (27%; Table 5.3). In terms of consumption of protein sources (without detailed nutritional analysis), the most commonly consumed source was tinned tuna recorded on 15% of all records, followed by fish (11%), shellfish (3%), terrestrial vertebrates (1.5%), livestock (1.5%: pigs (1%), chickens (0.5%)).

Table 5.2: The top five terrestrial plant and animal species consumed (from 24hr recall data only)

	Species		% frequency	% value (US$)	IUCN*
Plants	Sandpaper cabbage	*Ficus copiosa*	31.2	17.2	NA
	Tree fern	*Cyathea* spp.	19.2	8.6	NA
	Tahitian chestnut	*Inocarpus fagifer*	8.6	17.9	NA
	Coconut	*Cocos nucifera*	6.7	6.6	NA
	Breadfruit	*Artocarpus altilis*	3.4	6.8	NA
Animals	Coconut crab	*Birgus latro*	0.6	2.3	DD
	Northern cuscus	*Phalanger orientalis*	0.4	1.3	LC
	Prehensile-tailed skink	*Corucia zebrata*	0.2	0.3	DD
	Red-knobbed Imperial Pigeon	*Ducula rubricera*	0.1	0.2	NT
	Makira flying fox	*Pteropus cognatus*	0.1	0.1	EN

* IUCN RedList status: EN = endangered; NT = near-threatened; LC = least concern; DD = data deficient; NA = not assessed (IUCN 2014).

5.3.3 The effect of wealth and season on consumption of wild foods

The model with the highest Akaike weight, and therefore the best fit to the data, indicates the probability of wild food consumption to be influenced by household wealth, but not by season (Table 5.4), whereas Akaike weights for wild food

consumption models show this to be significantly shaped by both wealth and season (Table 5.4), with more wild foods consumed during the lean-season (5.8%) than non-lean season (4.3%). In both cases, the probability and quantity of consumption of wild foods was highest for the average-wealth households. Wealthy households managed to sustain consistent levels of consumption for all categories across both seasons, with a similar percentage composition of household consumption. During the lean-agricultural season, the composition of household consumption for poor and average-wealth households was marked by a decrease of 8.3% and 8.7% respectively for agricultural produce, and an increase in the consumption of wild foods (1.8% and 1.6% respectively) and imported foods (6.5% and 7.1% respectively; Table 5.3).

Table 5.3: Percentage composition of household consumption (standardized US$) for lean and non-lean seasons, for all households and each wealth category

Season	Wealth	Wild foods	Agriculture	Imported
Lean	All	5.8	65.3	28.9
	Poor	6.2	62.3	31.5
	Average	6.8	66.6	26.6
	Wealthy	4.4	67.0	28.6
Non-lean	All	4.3	71.4	26.6
	Poor	4.4	70.6	25.0
	Average	5.2	75.3	19.5
	Wealthy	3.4	68.2	28.4
All	All	5.1	68.3	26.6

Table 5.4: Results from both stages of the GLMM analysis for consumption of wild foods, ranked according to AIC weight. Model weights are also presented for null models for comparison. (N=776, households=72)

	Model	ΔAICi	Akaike weight
Probability wild food consumption (n=776, households=72)	wealth	0	0.51
	wealth+season	1.50	0.24
	wealth*season	2.26	0.16
	null	3.44	0.09
Amount of wild foods consumed (n=532, households=72)	wealth+season	0	0.95
	wealth*season	7.71	0.02
	null	6.86	0.03

5.3.4 The effect of wealth on sources of income

I found a highly significant positive association between wealth and sources of income (X^2=94.4, df=10, p<0.01). Wealthy households earned the majority of their income from cash crops (88%), followed by 62% for the average-wealth households and 23% for poor households (Table 5.5). Income from wild foods was low across all households, although wealthier households had the lowest income contribution from wild foods at 0.6%. However, in contrast to average and wealthy households, poor households derive the majority of their income (53%) from products derived from wild species. This was mainly earned through the sale of products woven from Pandanus leaves (57%), including sleeping mats and bags (typically produced by women), followed by carvings (42%) made from a variety of hardwoods (typically produced by men), see Table 5.6.

Table 5.5: Percentage of household income (standardized US$) by major sources for each wealth category

Wealth	Products derived from natural resources (from wild food)	Agriculture	Store	Cash crops
Poor	55.3 (2.4)	19.6	2.7	22.5
Average	28.7 (2.7)	5.6	4	61.7
Wealthy	7.3 (0.6)	3.7	1.5	87.6

Table 5.6: Sources of income (standardized US$) for products from natural resources

Product	**Producer**	**Species**		**% value (US$)**	**IUCN***
Sleeping mats; bags;	Women	Pandanus	*Pandanus tectorius;* *Pandanus* spp.	57	DD
Carvings (e.g. table, axe handle, bowl)	Men	Hard woods	*Pterocarpus indicus;* *Intsia bijuga;* *Pometia pinnata*	42	VU VU NA
Vine used in house building	Women/Men	Rattan	*Calamus hollrungii*	1	NA

* IUCN RedList status: VU = vulnerable; DD = data deficient; NA = not assessed (IUCN 2014)

5.4 Discussion

In this Chapter I aimed to assess the importance of wild foods in the household economy where availability is limited. Household wealth was found to influence the probability and consumption of wild foods, although the overall consumption of wild foods was relatively low. Household wealth was also found to influence sources of cash income, with wealthier households reliant on cash crops, unlike poor households who were heavily dependent on the sale of products derived from wild species.

5.4.1 The importance of wild foods where availability is limited

All households produced wild foods, with the vast majority of these used for consumption. Globally wild foods, particularly bushmeat, are a highly valuable commercial resource for rural communities (Milner-Gulland and Bennett 2003). However, because of the limited interaction with the cash economy there is an absence of a market for wild foods in Kahua, and consequently people are unprepared to pay for items they have access to themselves. Fish was the only wild food recorded as sold during this study, which may indicate that this resource is not always accessible to everyone, either through opportunity or equipment.

Wild foods contributed just 5% to total household consumption, which is considerably lower than reported for other studies. Even in global studies where wild food consumption was considered low, wild foods still contributed 18.9% and 15.4% of total household consumption of agricultural communities in the Democratic Republic of Congo (de Merode *et al.* 2004) and Tanzania (Powell *et al.* 2011) respectively. The particularly low consumption of wild foods in Kahua can be attributed to three key factors: 1) the naturally depauperate island environment (particularly mammals (McConkey and Drake 2006), but also plants (Whittaker and Fernández-Palacios 2007)); 2) the high opportunity cost of harvesting wild foods, resulting from high labour demands of other activities, such as cash crops. Labour demands of cash crops have also been associated with low bushmeat harvest in Ghana (Schulte-Herbrüggen *et al.* 2013), and high opportunity costs have also been linked to low wildlife harvest in other studies (e.g. Njiforti 1996; Wilkie and Carpenter 1999; Nielsen 2006); and 3) the presence of imported foods, which are viewed as superior to both agricultural and wild foods. An increasing preference for imported foods is often linked to the arrival of income generating activities which introduces a desire to generate income (Turner *et al.* 2007; Fazey *et al.* 2011). This leads to a social change with communities prioritising

activities, such as cash crops, which can generate disposable income to buy food, over subsistence activities which cannot earn extra income. This trend has been detected in Fiji, where an increase in income corresponded with a decline in fresh fish consumption replaced by purchased food (see Turner *et al.* 2007). This trend also appears to be occurring in Kahua, because I found imported foods contributed almost a quarter of total daily household consumption.

In this study, terrestrial plants were the most commonly consumed wild food comprising 86.6% of all wild food records. The most frequently consumed wild plants were leaves from *Ficus copiosa* and *Cyathea* spp., which were generally consumed as a meal accompaniment to starchy tubers, and used as an alternative to the main cultivated leafy green, *Abelmoschus manihot*. I consider the types of wild foods in the context of previous studies in the Pacific and wider Asian region. Consumption of wild plants were also found to be important to the Pwo Karen people in Thailand (Delang 2006). However, consumption values for wild plants have rarely been recorded for other studies in Asia-Pacific making relative comparisons of consumption difficult. Fish were the second most commonly consumed wild food (11% of records). Considering all communities were coastal, fish consumption was low compared to similarly isolated small island communities, such as Ahus Island (Papua New Guinea), where the majority of households listed fishing as a primary occupation (Cinner *et al.* 2005). However, other studies in coastal Melanesia have also found a tendency towards land-based livelihoods with a variable dependence on marine resources compared to the dominant role of agriculture which offers a more secure source of food (Polunin 1984; Turner *et al.* 2007). Terrestrial vertebrates accounted for just 1.5% of wild food consumption, which although low compared to other studies, is comparable to what would be expected according to availability. For example, a study in Papua (Indonesian New Guinea) found 51.1% of meals contained terrestrial vertebrates, with the most

frequently targeted mammals wild pig and rusa deer, followed by bandicoots (Pangau-Adam *et al.* 2012). In addition, a study in Sarawak found wild meat to be consumed in 6.3% of meals, with the low consumption attributed to overexploitation (Bennett 2002). The low consumption of terrestrial vertebrates in this study can be attributed to a low hunting success, which is a reflection of 1) the naturally depauperate environment, particularly the lack of large-bodied mammals, 2) overexploitation and decline of the few species that were present, and 3) inefficient hunting techniques (spear, bow and arrow, sling shot). Interestingly, wild food consumption was lower in Kahua than Papua, where hunters also used inefficient hunting techniques, and Sarawak, which is also an exploited environment. This suggests that it is the naturally depauperate environment which may be the most critical issue for the low consumption of wild foods, or indeed the cumulative effect of all three factors.

Despite the low consumption of wild foods, I found some increase in consumption of wild foods across all households during the lean-agricultural season. In line with other studies, this suggests that wild foods may provide a buffer against a shortage in agricultural food in Kahua (e.g. Brooks *et al.* 2008; León and Montiel 2008; Paumgarten and Shackleton 2011; Schulte-Herbrüggen *et al.* 2013). The use of wild foods during a shortage in consumption is consistent with a safety-net function, with wild foods providing a source of emergency sustenance at times of vulnerability (Shackleton *et al.* 2011).

5.4.2 The effects of household wealth on the use of natural resources

In line with many previous studies (e.g. Fisher 2004; Shackleton and Shackleton 2006; Fu *et al.* 2009), I found that the livelihoods of poorer households remain relatively more dependent on natural resources, with poor households earning the majority of their income (55%) from the sale of products derived from natural resources. The proportion

of income earned from natural resources in Kahua is higher than that found in other studies, which ranges from 20% in South Africa (Shackleton and Shackleton 2006), to 35% in Zimbabwe (Cavendish 2000) and 15-50% in rural southern India (Narendran *et al.* 2001). However, most studies have not disaggregated income by wealth, which could be obscuring higher values for the poor. Where household wealth has been considered, income source has been found to vary by wealth, but without a clear trend: poor households were found to benefit most from income from natural resources in Zambia (Kalaba *et al.* 2013) and Zimbabwe (Cavendish 2000), whereas average-wealth households were found to derive the greatest benefit in Cameroon (Ambrose-Oji 2003) and Nicaragua (Godoy *et al.* 1995). The divergent findings between studies may partly be attributed to different definitions of poverty, but are also likely to reflect the complexity of social and economic factors that determine differential access to natural resources within and between communities (de Merode *et al.* 2004). Wealthier households in Kahua earned the majority of their income from cash crops, indicating that wealthier households have access to more profitable and preferential income sources (cash crops) than forest products.

Household wealth was found to influence the probability and consumption of wild foods. Contrary to received wisdom, the poorest households are not always the most dependent on wild foods: in the Democratic Republic of Congo the value of wild foods was greatest for the wealthiest households (de Merode *et al.* 2004), and consumption of bushmeat was also found to increase with household wealth in Gabon (Wilkie *et al.* 2005). The results I found in Kahua are in concordance with a study by Godoy et al. (1995) who found that household dependency on wild foods was greatest for average-wealth households (Godoy *et al.* 1995). Godoy explained this quadratic relationship on the basis that poor households were unable to afford the necessary equipment to forage intensively whereas average households were able to make investments to

forage more, but once households become wealthier the forest became less economically significant. For Kahua, where poor households depend most on natural resources for their income, Godoy and colleagues' reasoning is unlikely to hold true. Instead, this relationship could be attributed to a relationship between access to land, labour and income-generating activities. Wealthier households in Kahua appear to have more access to land and labour which enables them to adequately balance production of food and income. In comparison, while average-wealth households also have access to land for cash-crops, the conflicting demands of household agriculture and cash-crop production mean that this may come at the expense of their food security. Cash income from the sale of cash crops is particularly unpredictable in Kahua because it is dependent on a ship servicing the island, which is irregular and dependent on good weather. Consequently, although households may have cash crops for sale there may not be a market present to sell them when needed. While wealthy households appear able to absorb these irregularities in market access and maintain their consumption, average-wealth households do not, and are forced to supplement their consumption with wild foods. Conversely, poor households who depend on locally-sold products derived from natural resources for their cash income are removed from external market demands, and mainly limited by the opportunity costs of harvesting which can be better controlled by the household. The lower involvement of poor households in cash cropping suggests that the poor have less access to such income sources, either through a lack of initial land holding assets and/or other assets such as education or labour. This is consistent with the view that the harvest of natural resources offers the best return for labour input where access to capital, land or livestock is limited (Shackleton and Shackleton 2006).

5.4.3 Implications for conservation

Local people in Kahua have reported declines in, and increasing travel time to harvest many natural resources. This suggests the current rate of exploitation is unsustainable, and with the rapidly increasing population, the pressure on natural resources can be expected to increase. Species which are prone to over-exploitation tend to have a restricted distribution, high habitat specificity and low reproductive rates (Sodhi *et al.* 2009). Worryingly, these are features of several species utilised in Kahua. For example, 75% of climbing *Pandanus* species found in the Solomon Islands are endemic (Pacific Horizon Consultancy Group 2008), and population declines have already been detected for the coconut crab (*Birgus latro*) throughout its range (Brown and Fielder 1991; Lavery *et al.* 1996). In addition, the two largest species of flying fox (*Pteropus* spp.) on Makira were found to be consumed. Bats are the only native mammals on Makira and consequently potential keystone species in the ecosystem as pollinators and seed dispersers (Cox *et al.* 1992; Mickleburgh *et al.* 2009). Depauperate island ecosystems have less functional redundancy than larger continental areas, and the over-exploitation of certain species, such as flying foxes and large-bodied pigeons (e.g. *Ducula* spp.) could result in the loss of ecological functions, such as seed dispersal (McConkey and Drake 2006). This in turn could affect plant community composition (Christian 2001), for example, by shifting forest composition in favour of small-seeded trees. Moreover, the primary seed dispersers of *Pandanus* spp. are large flying foxes (Fujita and Tuttle 1991) highlighting a possible positive feedback loop between the consumption of flying foxes and the income generating opportunities of the poor.

Overharvesting of natural resources is a major threat to biodiversity in the humid tropics (Milner-Gulland and Bennett 2003), but if people are to be cautioned against harvesting keystone species to ensure the continuation of the ecological role these

taxa provide then alternative protein sources need to be found. Many tropical forest people already rear livestock, but for reasons of taste, cost, or culture, they are generally kept as insurance for sale during emergencies, and only consumed at ceremonial or other special occasions rather than daily subsistence (Bennett 2002). Indeed, in Kahua I found more terrestrial vertebrates consumed (1.5% of records) than either pigs (1%) or chickens (0.5%). Although considerable effort can be required to access and capture bushmeat, the cost of hunting an open access resource is invariably lower than raising pigs or chickens (Kaschula and Shackleton 2009). In this study, tinned tuna was the most commonly consumed protein source, and there is a general trend for a preference of imported foods over traditional foods. The transition to imported foods may benefit conservation, because it provides an alternative protein source to bushmeat and its availability is likely to increase in the future. However, while the increase in consumption of imported foods may be a good thing from a conservation perspective, it may not benefit long term development. Increasing consumption of imported foods increases vulnerability in the food supply, particularly during hazards (given low incomes and relatively high food prices; Barnett 2011), and also reflects the 'nutrition transition' to cheap, poor quality foods of little nutritional value. This transition has been identified as a contributing factor to increasing rates of non-communicable diseases such as obesity, diabetes and heart disease across the Pacific (Barnett 2011), and other developing countries around the world (Popkin 2004). Identifying alternative food sources which are culturally-appropriate, nutritious, and sustainable is a global priority which cannot wait until wildlife populations have dropped to unrecoverable levels. Tackling this multidisciplinary issue of biodiversity conservation and food security will require the full integration of conservation and development agendas at local, national and international levels.

So far in this thesis, I have established an appropriate measure of household wealth and determined that poor households are more dependent natural resources for their cash income, whereas wealthier households rely on cash crops. But I do not yet know how these different livelihood strategies are impacting the ecological system. In the next Chapter, I assess how different land use intensities, associated with these different livelihood strategies, are impacting biodiversity.

Chapter 6: Impacts of land use change on tropical avifauna

Impacts of land use change on tropical avifauna

Material from Chapter 6 formed the basis of the manuscript, which has been submitted as: *Davies T.E., Clarke, R.H., Ewen, J.G., Fazey I.R.A., Pettorelli, N., Cresswell W. Impacts of land use change on endemic avifauna on Makira, Solomon Islands: endemics avoid monoculture. Submitted to EMU*

View of Hanimanu village, west Wainoni

6.0 Chapter Overview

In this Chapter, I now turn to the ecological system to assess how biodiversity is changing with increasing intensity of land use across the five dominant land uses in Kahua associated with different livelihood strategies. I use birds as my focal taxa because they have been identified as excellent indicators of biodiversity and land use change. In Chapter 2, I identified a multi-dimensional measure of biodiversity to be important. Therefore in this Chapter, I go beyond simple measures of species richness, and assess avian species richness, proportion of endemic species, community composition, and functional traits.

6.1 Introduction

Tropical forests cover just 7% of the Earth's land surface, but support more than 60% of known species (Laurance 1999; Wright 2005; Bradshaw *et al.* 2009). These forests provide essential ecosystem services, such as carbon storage, clean water provision and contributions to climate stability, playing a major role in human well-being (Tobias *et al.* 2013). However, deforestation is continuing at unprecedented rates with 1.2% of total rainforest area lost each year (equivalent to >15 million hectares per year, Laurance 1999). This habitat loss is largely driven by intensification of land use, which has led to fragmentation and degradation of habitat (Swift and Hannon 2010). Habitat loss is particularly rapid in developing regions where human populations are expanding and where the majority of people depend on natural resources for subsistence (Msuha *et al.* 2012). Population growth rates in these regions are substantially higher than global rates (Cincotta *et al.* 2000). With consumption levels expected to exceed population growth (Sodhi *et al.* 2013), agricultural expansion will remain a major factor in land conversion in the tropics and a significant driver of biodiversity loss.

Tropical islands are an important focus for conservation efforts. They are experiencing proportionally higher rates of deforestation than continental areas (Achard *et al.* 2002) and support high numbers of endemic species with traits that make them particularly prone to extinction, such as habitat and diet specialisation, and a restricted range (Purvis *et al.* 2000; Aratrakorn *et al.* 2006; Senior *et al.* 2013). Understanding how island endemic species respond to land use change, and the degree to which tropical forest organisms can persist in human-dominated landscapes can improve decision making for environmental management (Flynn *et al.* 2009; Gardner *et al.* 2010). However, the impacts of land use change on tropical island biodiversity have been little-studied. Consequently, the conservation benefits of modified landscapes in these regions are poorly understood (Goldman *et al.* 2008; Sodhi *et al.* 2010; Woinarski 2010). This is of particular concern given the disproportionately high number of endemic species that are threatened with extinction driven by land use change (Woinarski 2010; Waltert *et al.* 2011; de Lima *et al.* 2012).

As a taxonomic group, birds are an exemplar for assessing how species' traits lead to extinction through the effects of land use change because they are widely distributed and occupy a broad range of habitat types and ecological niches, they are also well studied with well-resolved taxonomy (Barlow *et al.* 2007; Gardner *et al.* 2008; Kessler *et al.* 2011). Birds are generally easy to identify in the field and survey techniques for birds are comparatively simple and cost-effective (Gregory *et al.* 2005). Birds also have vital roles in key ecosystem processes, such as pollination, seed dispersal, nutrient cycling, pest control and scavenging (Whelan *et al.* 2008; Wenny *et al.* 2011), and provide a link that connects habitats and ecosystems through their movements (Lundberg and Moberg 2003). Thus, bird populations are useful for monitoring structural and functional changes in ecosystems (Hausner *et al.* 2003), with declines in bird populations potentially leading to changes that cascade through ecosystems and

subsequently cause declines in benefits to humans (Wenny *et al.* 2011). Anthropogenic modification of vegetation alters habitat structure and resource provision, which may change the interactions between fauna, favouring some species and disadvantaging others. Traits, such as body size and feeding guild, group species according to shared responses to environmental disturbances and are often strongly associated with functional traits, which classify species based on their shared effects on particular ecosystem functions (Lavorel and Garnier 2002). Consequently, assessing how different functional traits are affected by land use change can reveal concurrent impacts on ecosystem function (Hooper *et al.* 2005; Senior *et al.* 2013) and aid conservation efforts by predicting species groups at greatest risk from land use change (McGill *et al.* 2006; Williams *et al.* 2010).

In this study, I assess the impacts of land use change on avifauna in a tropical island system using Makira, in the Solomon Islands, as a case study. The Solomon Islands, located east of New Guinea in the South West Pacific, present an ideal study site as they are part of the East Melanesian Islands biodiversity hotspot (Myers *et al.* 2000) and contain one of the last remaining tracts of undisturbed coastal tropical rainforest (Bayliss-Smith *et al.* 2003). However, these islands are undergoing rapid land use change due to an economy strongly biased towards extractive industries, , primarily logging (CBSI 2011), coupled with one of the highest human population growth rates in the world (2.6% per annum; UNICEF 2011). Despite a looming biodiversity crisis, the Solomon Islands remain severely understudied. I examined bird species richness and community assemblages, with a particular focus on endemism and functional traits, to provide further insight into the 'conservation value' of the five dominant habitats present on Makira (Barlow *et al.* 2011; Waltert *et al.* 2011; de Lima *et al.* 2012; Senior *et al.* 2013). Specifically I aimed to quantify the impacts of land-use change on the Makira avifauna by assessing variation, along a gradient of human disturbance, in: 1) species

richness, 2) community composition, including endemism and functional traits (microhabitat and feeding guild), 3) species-specific changes in abundance (community structure).

6.2 Methods

6.2.1 Study area

As detailed in Chapter 3, this study was focussed in the Kahua region of Makira (Figure 3.1) in the Solomon Islands. Makira contains globally significant biodiversity and high levels of endemism; it is part of a high priority Endemic Bird Area (Birdlife International, 2013). Makira, with 18 extant endemic bird taxa, holds more restricted-range bird species and subspecies than any other area of comparable size in the world (Danielsen *et al.* 2010).

6.2.2 Data collection

Data were collected between January and July 2012. Two 500 m length transects were established in each of the five land-use types (N = 10; Figure 6.1; Appendix A9). Line transects are a simple way to survey bird populations. Unlike traditional point counts, where data are collected from a specific point on the landscape, line transects collect continuous data over the course of a 'walking transect'. This data collection method is particularly useful for species that are not easily detected through point counts either because they are found locally, they are rare or uncommon, or because they are clustered (Lloyd *et al.* 1998; Gregory *et al.* 2004). In order to improve the collection of data on rare or uncommon species and to collect the most robust data set, I decided to use line transects. The sampled land use types increased in intensity of land use and were classified as:

1) Intact forest: closed canopy (30-45 m high), comprising large, hardwood trees, including those of higher quality timber (*Pometia pinnata, Vitex cofussus, Pterocarpus indicus, Calophyllum vitiense*), with dense understory vegetation including thickets of smaller trees, rattan palms (*Calamus* spp.), *Stenochlaena* ferns and *Selaginella* mosses. Anthropogenic disturbance is a ubiquitous feature of the forests of the Solomon Islands (Bayliss-Smith *et al.* 2003), as such no forests in this region can be considered "primary" in its truest sense. I therefore use intact forest to refer to the lowland evergreen tropical rainforest with historical but presently limited human disturbance.

2) Secondary forest: no continuous canopy, although crowns can be in close proximity to one another, mainly composed of small fast growing, pioneer species, (including *Macaranga* spp*., Ficus* spp*.,* and *Hibiscus tiliaceus*) interspersed with larger trees, including Ngali nut (*Canarium indicum*), breadfruit (*Artocarpus altilis*), coconut (*Cocos nucifera*) and sago palm (*Metroxylon salomonense*). This habitat is often used intensively by local communities for the collection of firewood, timber and wild plants.

3) Garden: an open canopy above food crops such as yam (*Dioscorea* spp.), taro (*Colocasia esculenta*), sweet potato (*Ipomoea batatas*) and slippery cabbage (*Abelmoschus manihot*)*,* as well as various protected or deliberately planted herbaceous and tree species, such as coconut palms, banana cultivars (*Musa cultivars*), sago palm, betel-nut palm (*Areca catechu*), nut trees (*e.g*. *Canarium* spp., *Barringtonia edulis*, *Inocarpus fagifer)*, edible figs (*Ficus* spp.), and fruit trees (e.g. *Atrocarpus altilis , Mangifera indica*, *Carica papaya*).

4) Mixed-cocoa: smallholder plantations of the cocoa tree (*Theobroma cacao*) which typically grow to heights of 4-8 m high*.* Cocoa trees are planted close together resulting

in a low closed canopy with no understory due to regular clearance. Small patches of cocoa trees are regularly interspersed with coconut palms and large fruit and nut trees, such as breadfruit and *Canarium* spp, with irregular patches of low-canopy regenerative tree and herbaceous species, including gingers (Zingiberaceae) and betel-nut.

5) Monoculture cocoa: extensive smallholder cocoa plantations, grown in flood plain areas with a 4-8 m high closed canopy of low height and no understory (because it is regularly cleared). Monoculture cocoa is irregularly interspersed with coconut trees and occasionally large lone trees, (1-2 per transect) such as breadfruit or Malay apple (*Syzygium malaccense*), but with almost no shade cover.

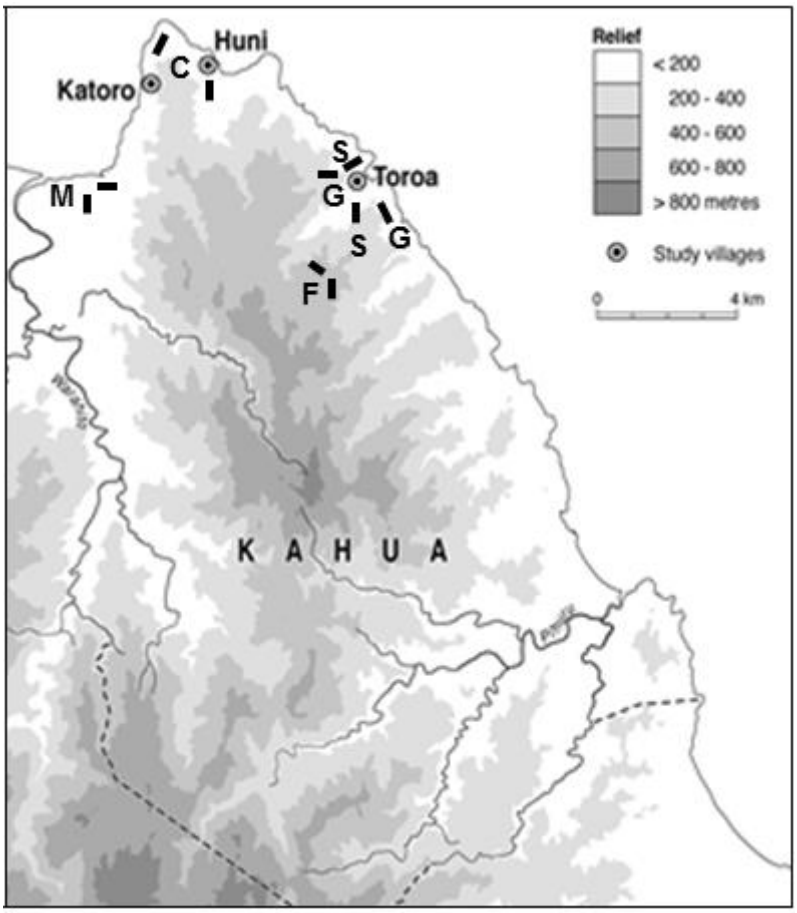

Figure 6.1: Location of Kahua study site, and location of the ten transects in the different habitats (F= forest, S= secondary, G= garden, C= mixed cocoa, M = monoculture cocoa).

Transects were traversed on foot. Every bird that was detected by sight or sound was recorded. The perpendicular distance from the transect line was estimated at the time of detection (Bibby *et al.* 2000), by walking up to the point on the transect that was deemed perpendicular to the bird and the distance estimated from this point. Calling birds were recorded as if visual sightings although extra care was taken in estimating the distance from the observer. To maximize detection, surveys were conducted between 06:00 – 08:00, usually within 30 minutes of sunrise; in appropriate weather conditions (e.g. surveys were not conducted in heavy rain and/or high wind). Transects were walked at a steady pace to avoid double counting of birds. Individual transects

took no more than 40 minutes to complete. Topographically, all transects were situated between 20 m and 500 m above sea level to avoid habitat changes associated with higher altitudes. All transects were completed by myself and four local assistants throughout the survey period. Prior to data collection, a significant training period in call identification was undertaken. This included listening to recordings from Xeno-canto (www.xeno-canto.org), spending time in the field listening to birds and speaking to local people about the birds and their calls. Experience made bird identification problems minimal. Distance estimation was also practiced during this training period, including estimating distances to calling birds and then subsequently determining the actual distance by pacing. This approach has been shown to considerably improve the accuracy of distance estimates (Reynolds *et al.* 1980). Only one transect was completed per day. Transects were typically repeated once per fortnight and each was surveyed at least monthly. Transects within the same land use type were not surveyed in the same fortnight period and were as far apart from each other as the terrain allowed (range 10 m – 2 km). Each transect was repeated a minimum of six times, with a minimum of 13 surveys conducted in each habitat type.

Prior to analysis, I excluded species that were in-flight and not directly interacting with the habitat. The sole exception to this was the Pacific swallow (*Hirundo tahitica subfusca*) which forages just above the uppermost vegetation strata and thus targets flying insects directly associated with the habitat of interest. Swiftlets (*Collocalia esculenta*, *Aerodramus spodiopygius*, *A. v. vanikorensis*) were excluded as they forage much higher above the uppermost vegetation strata and their presence is more likely to be driven by broader-scale vegetation structure than the scale of the transect.

In most bird surveys, not all species that are actually present are recorded (see Nichols and Conroy 1996). The detectability of birds is typically lower in densely vegetated

forests with tall inaccessible canopies, such as intact forest, compared to more open modified habitats with broken, lower canopies, such as monoculture cocoa (Gardner *et al.* 2009). Although data were collected in a format suitable for density estimation controlling for detectability variation using the standard Distance methodology (Thomas *et al.* 2010), accurate detectability functions require approximately 60 records per species per detectability class (in this case the five habitat classes). Very few species were recorded sufficiently frequently to reach this target so that application of the Distance method would have generated density estimates with sufficiently large confidence limits that no meaningful comparisons between habitats could be carried out. I therefore adopted an approach to minimise the effects of detectability and to establish whether detectability effects may have biased (or confounded) raw count comparisons across the habitats.

First, I minimised the effects of differences in detectability within my data by determining the maximum appropriate transect width as 50 m on either side of the transect line (99% of all records were obtained within these bounds) and excluded the small proportion of records obtained at distances >50 m. Second I established how detectability varied across habitat. Even if abundance varies across habitats, the relative frequency distribution of sightings in particular distance bands will be the same if habitats have the same detectability. For example in a dense habitat most sightings might be at a few meters from the observer, whereas in an open habitat they might be evenly spread across distance bands. Significant differences in the relative distribution of detectability distances across habitats can then be assessed using Chi-square tests of distance band by habitat. I applied Chi-squared tests to the distance data (divided into bands: <10 m, 11-50 and in-flight) across habitats for the 17 most abundant species (those with >50 records). Eight of the 17 (47%) species tested were found to differ significantly across the distance bands suggesting that detectability might

confound raw count comparisons across habitats. In one species, the Chestnut-bellied Imperial Pigeon, detectability differences across habitats arose because of in-flight records and so all in-flight records for this species were removed from the dataset before further analysis. Of the remaining seven species with variable detectability across habitats, four (Makira Cicadabird, Red-knobbed Pigeon, Oriole Whistler and Yellow-bibbed Fruit Dove) higher numbers were recorded in more densely vegetated habitats where detectability would be expected to be lowest: therefore the conclusion that the species were more common in these habitats did not arise because they were more detectable there.

6.2.3 Data analysis

All analyses were conducted using R (version 3.0.1; R Core Team 2013) and focussed on presence-absence and total count data at the habitat level. I calculated observed species richness (the total number of species recorded) and a Shannon-weiner index for each habitat type. I used EstimateS v9.0 (Colwell 2013) to calculate first-order jackknife, to estimate the total species richness of each land use, and built sample based rarefaction curves to compare species richness across land uses (Gotelli and Colwell 2001). I collated information on endemism (Makira, Melanesia, or widely distributed) using Dutson (2011) and functional traits (microhabitat selection: water, aerial, canopy, canopy/understory, understory, or terrestrial; and foraging guild: fruit and other vegetative material, nectarivore, omnivore, insectivore or vertebrates) using the best available published material from a study in another area of the Solomon islands by Kratter *et al*. (2001) and my own extensive field experience (Appendix A9).

I first visualised the similarities in avian community composition between land uses using the vegan package to perform a cluster analysis to assess community similarity (Oksanen *et al.* 2013). I used Chi-squared tests to assess the differences in

proportions of species between habitats for both microhabitat selection and foraging guild. To model the difference in abundance of each species individually and from that inferring differences in community composition, I used a multivariate generalised linear model on species count data using the mvabund package (Wang *et al.* 2012). I included habitat (categorical: intact forest, secondary, garden, mixed-cocoa, monoculture) and survey date as predictor variables. Date was included to explore the possibility of seasonal differences. However, as no support was found for seasonal effects ($p = 0.187$), it was removed from the model (Crawley 2007). Multivariate generalised linear models provide a powerful framework for analysing species abundance data and have been shown to be more robust than distance based methods, such as multidimensional scaling and redundancy analysis (Warton 2011; Warton *et al.* 2012; Sreekar *et al.* 2013). I specified a negative binomial error structure in the model because the count data was over-dispersed (O'Hara and Kotze 2010; Wang *et al.* 2012).

6.3 Results

I recorded a total of 3601 individuals from 42 bird species, including 12 of the 18 (67%) extant endemic species on Makira (Appendix A10). Those that were not recorded included Shade Bush Warbler (*Cettia parens*), Island Leaf Warbler (*Phylloscopus poliocephalus makirensis*), Dusky Fantail (*Rhipidura tenebrous*) and Grey-throated White-eye (*Zosterops ugiensis ugiensis*), all of which are known to be found at higher elevations (>500 m) than sampled in my study. I also had no records of the nocturnal Makira Boobook (*Ninox roseoaxillaris*) which is considered rare (Dutson 2011), nor Makira Moorhen (*Gallinula silvestris*) which is listed as Critically Endangered (BirdLife International 2013).

The observed number of species found in each habitat ranged from 26 species in monoculture cocoa to 34 species in secondary forest (Table 6.1). Rarefaction curves revealed that species richness had not yet completely plateaued in any of the habitats, and therefore Jackknife species richness estimators were higher than the observed species richness. This is the case for many surveys in tropical habitats (Gotelli and Colwell 2001). My observed species richness is therefore an underestimate of the actual number of species present. Shannon-weiner index ranged from 2.66 for intact forest to 2.93 for mixed-cocoa.

Table 6.1: Summary statistics for avian species richness and assemblage metrics within the five dominant habitats on Makira, Solomon Islands

Habitat	Species richness	Shannon-weiner	Number of Makira endemic sp.	Percentage of Makira endemic sp. (%)	Number of Melanesia restricted sp.	Percentage of Melanesia restricted sp. (%)
Forest	27	2.66	11	36	10	54
Secondary	34	2.88	11	27	11	28
Garden	30	2.87	9	37	11	38
Mixed cocoa	33	2.93	11	30	9	33
Monoculture	26	2.72	7	28	8	20

Eleven endemic species were recorded in intact forest, as well as in secondary and mixed cocoa habitats, with the lowest number of endemics (7) found in the monoculture cocoa. The number of Melanesia restricted range species by habitat increased slightly with decreasing levels of agricultural intensification (secondary forest = 11 species, garden = 11, intact forest = 10, mixed cocoa = 9, monoculture cocoa = 8).

A cluster dendrogram illustrated that there were differences in community composition across land uses, but offers little information on the nature of these differences. Therefore I further explored the differences using more nuanced methods.

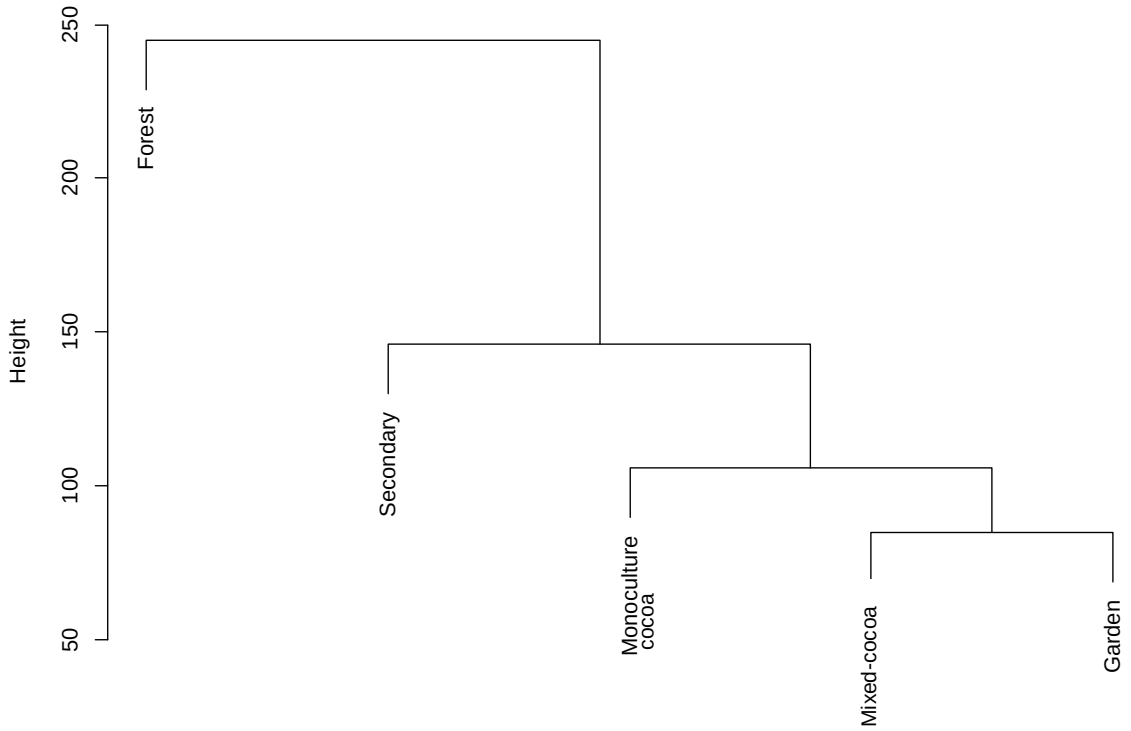

Figure 6.2: Cluster dendrogram to visualise similarities in avian community composition across land uses.

The relative distribution of functional traits displayed by birds differed between habitats, both in terms of microhabitat selection (X^2= 141.2, df = 20, p <0.01) and foraging guild (X^2=235.1, df = 16, p <0.01). Monoculture cocoa had the highest percentage of canopy species (71.7%), followed by mixed cocoa (64%) and garden (58.6%). Intact forest contained the highest proportion of canopy/understory species (43.3%) followed by garden (39.2%). The highest proportion of terrestrial species were recorded in monoculture cocoa (4.3%) and mixed cocoa (2.7%), with the highest proportion of understory species recorded in forest areas (1.1%). Secondary forest contained the highest percentage of frugivores (56.9%), followed by garden (45.4%) and mixed cocoa (38.2%). Garden areas contained the highest percentage of insectivores (40.6%), followed by intact forest (39.2%). Intact forest areas contained the highest percentage

composition of omnivorous birds (8.6%) followed by garden (4.2%). The highest percentage composition of nectarivorous birds was found in monoculture cocoa (22.2%) followed by mixed cocoa (16.1%) and intact forest (15.3%).

I found significant differences in species abundance between habitats (Dev = 800.2, df = 71, p=0.001), with my data suggesting 15 species had a distinct preference for one or more specific habitats (Table 6.2, Figure 6.3). Of these species, 80 % were Makira endemics (6) and Melanesia endemics (6). Over half of all species found to vary by habitat (7/15) were absent from the most heavily disturbed habitat. An assessment of the mean number of these species seen per transect indicates that some species are habitat specialists, for example Barred Cuckooshrike, Cardinal Lory, Makira Cicadabird, Makira Honeyeater, Oriole whistler and Yellow-bibbed Lory can be considered forest specialists with only limited intrusion into other habitats (Figure 6.3). Conversely, Coconut Lorikeet and Collared Kingfisher appear to benefit from agricultural intensification as they only occupied modified habitats. In addition, species such as Chestnut-bellied Monarch, Mackinlay's Cuckoo-dove and White-headed Fruit-dove appear to preferentially target habitats with intermediate levels of disturbance (Figure 6.3).

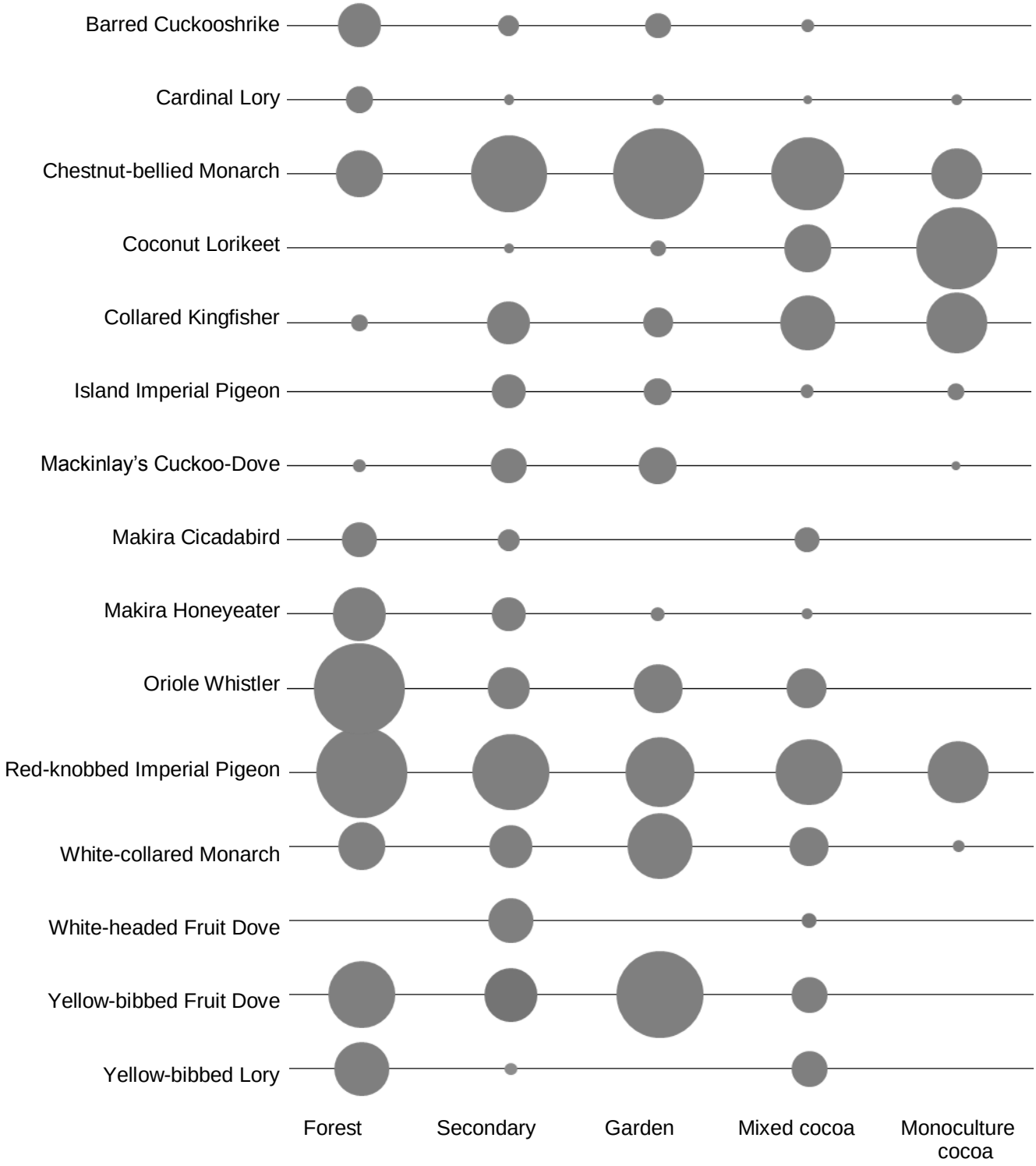

Figure 6.3: Relative abundance across habitats for species which showed significant differences in their distribution. Abundance of each species in the different land uses corresponds to the size of the circle.

Table 6.2: Variation of avian species composition between habitats on Makira, Solomon Islands (mvabund: Dev = 800.2, df = 71, p=0.001). Species that varied in abundance across habitats are denoted with '*'alongside the habitat in which they were found to be most abundant.

Species	Dev	Pr(>Dev)	*N* records	Most abundant in:
Barred Cuckooshrike*	22.1	0.004	36	Forest
Cardinal Lory*	18.7	0.024	15	Forest
Chestnut-bellied Monarch*	26.4	0.001	236	Garden
Coconut Lorikeet*	36.8	0.001	94	Monoculture
Collared Kingfisher*	28.1	0.001	100	Monoculture
Island Imperial Pigeon*	20.9	0.010	29	Secondary
Mackinlay's Cuckoo Dove*	36.1	0.001	35	Secondary
Makira Cicadabird*	25.1	0.002	55	Forest
Makira Honeyeater*	83.7	0.001	70	Forest
Oriole Whistler*	87.6	0.001	228	Forest
Red-knobbed Imperial Pigeon*	25.5	0.002	222	Forest
White-collared Monarch*	24.3	0.002	77	Garden
White-headed Fruit Dove*	22.8	0.002	33	Garden
Yellow-bibbed Fruit Dove*	70.3	0.001	124	Forest/Garden
Yellow-bibbed Lory*	32.1	0.001	43	Forest
Beach kingfisher	8.0	0.53	3	
Brahminy Kite	3.5	0.72	1	
Bronze Ground Dove	7.8	0.56	5	
Buff-banded Rail	4.0	0.72	2	
Cardinal Myzomela	14.3	0.11	75	
Chestnut-bellied Imperial Pigeon	16.0	0.06	65	
Common Kingfisher	7.5	0.56	4	
Crested Cuckoo-dove	11.7	0.28	13	
Dollarbird	9.4	0.41	16	
Eclectus Parrot	1.2	0.89	39	
Long-tailed Triller	6.9	0.56	58	
Makira Starling	3.8	0.72	10	
Melanesian Megapode	11.0	0.29	16	
Metallic Starling	7.9	0.54	270	
Mottled Flowerpecker	8.9	0.45	94	
Nankeen Night Heron	6.8	0.56	3	
Pacific Baza	3.2	0.73	3	
Pacific Koel	9.2	0.41	4	
Pacific Swallow	8.5	0.49	12	
Pale-vented Bush Hen	9.4	0.41	3	
Pied Goshawk	12.8	0.19	12	
Rufous Fantail	14.9	0.08	252	
Singing Starling	7.6	0.56	146	
Sooty Myzomela	10.6	0.29	183	
Stephen's Emerald Dove	6.2	0.56	14	
Variable Dwarf Kingfisher	15.2	0.08	13	

Willie Wagtail	13.6	0.14	19

6.4 Discussion

I assessed the impact of land use change on the avifauna of Makira, a tropical island system, and found species composition varied with land use. Endemic and restricted range bird species appear to be particularly susceptible to increasing intensity of anthropogenic land use as many of these species are forest specialists. As tropical islands have less functional redundancy than continental areas (McConkey and Drake 2006), protecting the remaining forest will be critical to conserve these endemic species and maintain ecosystem functioning.

6.4.1 Changes in species richness

I found species richness to be similar across habitats, but that endemic species richness tended to decrease in more intensive land uses. The trend in species richness across land use intensity, with higher species richness in lightly disturbed environment of secondary forest is indicative of the intermediate disturbance hypothesis (Connell 1978). A similar trend has been observed in Cameroon, where forest-dependent species were replaced by species typical of open habitats, resulting in stable species richness but increasingly homogenous assemblages across habitats (Lawton *et al.* 1998). Although species richness is a commonly used metric of biodiversity, it provides only a coarse index with no information on abundance, or differentiation between vulnerable and disturbance-adapted species (Laurance *et al.* 2006; Pardini *et al.* 2009). My findings that endemic species richness declines with increasing intensity of land use is consistent with other studies (McKinney and Lockwood 1999; de Lima *et al.* 2012), and supports endemic species richness as a useful metric when assessing the conservation value of modified habitats (Waltert *et al.* 2011).

6.4.2 Changes in species composition

I found significant differences in the relative distribution of functional traits displayed by birds between habitats, including a proportional increase in nectarivores and decrease in insectivores in the most disturbed habitat, monoculture cocoa. Similar responses by nectarivores have been observed in Borneo (Cleary *et al.* 2007), New Guinea (Marsden and Symes 2008), and the wider tropics (Waltert *et al.* 2005; Tscharntke *et al.* 2008). I also observed a proportional increase in frugivores in intermediately disturbed areas of secondary and garden habitats. In contrast, insectivores and frugivores have been found to consistently decline as intensity of land use increases (Marsden and Symes 2008; Newbold *et al.* 2013). Elsewhere this has been attributed to a reduced abundance of fruiting trees and invertebrates (Cleary *et al.* 2007; Vetter *et al.* 2011). That the traditional agroforestry system in the Solomon Islands (and across the wider Pacific region) generally preserves fruiting trees throughout the landscape (Mertz *et al.* 2012) may explain my contrasting results, as even isolated fruiting trees in agricultural areas have been shown to provide important resources for frugivorous birds (Şekercioğlu *et al.* 2007). The change in feeding guilds I observed suggests bottom-up restrictions such as habitat and resource limitations, such as food availability.

6.4.3 Changes in species abundance

I found significant differences in species assemblages, with a suite of species showing distinct habitat preferences. Such changes in abundance are known to have important effects on the structure of communities and on the functioning of ecosystems (Gaston and Fuller 2008). A high proportion of the species that were found to differ across habitats were endemic and Melanesia restricted range species. The endemic Barred Cuckoo-shrike, Makira Honeyeater, Makira Cicadabird, and restricted range Cardinal Lory and Yellow-bibbed Lory appear to be forest specialists with only limited intrusion

into other habitats. These species are likely to be sensitive to increasing intensity of land use. In contrast, the Oriole Whistler is a forest species that appears to be able to occupy modified habitats, although the extent to which it is able to complete its life cycle in such habitats is unknown.

The White-collared Monarch and Chestnut-bellied Monarch appear to be disturbance specialists as they were found to occur at higher densities in areas of intermediate disturbance. This may be because of increased feeding opportunities in lightly-disturbed environments, relative to heavily-disturbed environments where a decrease in plant diversity is tracked by a corresponding decrease in invertebrate diversity (Bennett and Gratton 2013). Although fewer than 1% of the world's birds species primarily prefer agricultural areas, nearly one-third of all birds occasionally use such habitats (Şekercioğlu *et al.* 2007). Persistence of these species in these areas likely depends on the configuration of forested and non-forested patches (Tscharntke *et al.* 2008), with metapopulation theory suggesting that populations can be maintained in lower quality habitat patches by influxes of individuals from source populations in high quality habitat. In this context, further loss of intact forest or increased habitat fragmentation will have increasingly negative effects on the native avifauna on Makira. Furthermore, extinctions of forest birds often only appear after considerable time-lags (see Brooks *et al.* 1999), which can only be detected through long-term monitoring of population dynamics (Waltert *et al.* 2005).

I found two species that showed a preference to heavily disturbed environments. The Collared Kingfisher and Coconut Lorikeet were recorded in all habitats, but displayed a trend towards higher numbers in heavily disturbed habitats especially cocoa monocultures. Consistent with their ability to exploit human modified habitats both of

these species are of lower conservation concern (BirdLife International 2012b; BirdLife International 2012a).

6.4.4 Limitations of the current study

Results of studies like this one can be biased by issues of detectability (Thornton *et al.* 2011). Although I found certain species to be more detectable in some habitats than others, 60% of these species were recorded in lower numbers in more open habitats suggesting a true absence in these areas. While it is likely that some species that were present went undetected, and many individuals of species were not recorded in the denser habitats; my findings in relation to decreasing numbers of endemics with increasing land use intensification are conservative given these same habitats are also more open, and where species are more detectable. The true abundance of these species may be much greater in dense habitats, and thus intact forest areas may actually be even more important for these species than I suggest here. More robust density estimates would be obtained by strict application of Distance methodology (Thomas *et al.* 2010) but such methods require large amounts of data to obtain reliable detectability functions and several of its key assumptions are violated in rainforest situations.

Due to the mosaic of land use and extreme terrain, some of the transects were positioned close together, possibly creating a problem of spatial-autocorrelation. I attempted to minimise any effects of non-independence by not surveying transects in close proximity within the same two-week period. However, this study would be improved through fully-independent transects and sampling across a wider area. In addition, I did not account for habitat area or proximity to other habitats, which could influence bird communities. Other studies have found proximity to intact forest to be an important influence on the proportion of frugivores and insectivores present (Marsden

and Symes 2008). An improvement in study design could be achieved through the use of remote sensing data to map land uses, and also to assess rates of land use change (see Buchanan *et al.* 2008). However, this type of data was not available for Kahua, which is difficult to study using weather-dependent remote sensing data due to year-round abundant rainfall, high humidity and high altitude areas, resulting in significant cloud contamination. Ideally, multiple uncontaminated very high resolution images and LIDAR technology would be the best option to monitor land use change in this region (Goetz *et al.*2009), but to my knowledge no such data currently exists in the public domain (see Garonna *et al.* 2009).

No single species or taxon can be expected to adequately represent or indicate patterns for all other species or taxa (Pearson 1994). Although birds have been shown to be excellent indictors, they are highly mobile and the responses of birds may not accurately reflect the responses of more sedentary, spatially-restricted faunal taxa such as invertebrates, reptiles and small mammals (Gregory *et al.* 2005). Therefore, the robustness of my findings would be improved through the collection of data across multiple taxa to determine if avian responses to land use change are congruent with other taxa (Howard *et al.* 1998).

My findings are from a small area of Makira, and therefore broader inferences should be conservative. However, the Kahua region is one of the only areas on Makira not to have experienced commercial logging and as such the Kahua region may represent a more intact avifauna than other areas of Makira.

6.4.5 Implications of future land use change

My findings are consistent with other studies that suggest replacing forests with more intensive land uses results in a shift toward bird communities with less specialized traits

and a reduction in the diversity for some key functional groups (Şekercioğlu 2012; Luck *et al.* 2013). Land use change presents a serious threat to forest birds, especially for species with certain functional traits (Newbold *et al.* 2013). This trend is a particular concern for tropical islands because not only do islands have endemic fauna which is more prone to extinction (88% of bird extinctions since 1600 have occurred on islands, Butchart *et al.* 2006), they also have less functional redundancy (Wardle 2002; McConkey and Drake 2006). Thus, the loss of island species that fill key functional roles may inordinately disrupt vital ecosystem services and processes, making them less resilient to other environmental changes (Hooper *et al.* 2005). In areas where the human population relies heavily on forest products (as is the case on Makira) such disruption will likely lead to social and economic vulnerability, and ultimately degraded socio-ecological systems (Elmqvist *et al.* 2003).

Shifting cultivation has occurred on the Solomon Islands for at least 3000 years (Mertz *et al.* 2012) and consequently it is likely that the most sensitive species became extinct a long time ago (e.g. large flightless birds that were easy to catch with dogs). Therefore, the remaining fauna consist of species that are already adapted to some level of human disturbance, from low disturbance intact forest, to high disturbance plantations. However, the recent introduction of cash crops has promoted a shift towards more intensive land uses. Avifauna assemblages appear most negatively affected by this intensive land use, with many species of conservation value not found in this habitat. Currently, these monocultures are spatially constrained and cover relatively small areas in Kahua, but should their development continue on the current trajectory, negative repercussions for biodiversity and ecosystem function would appear inevitable.

6.4.6 Implications for conservation

Despite high levels of endemism on Makira and across the Solomon Islands, relatively little is known about the distribution, ecology and threats affecting many of the endemic species. Of the endemic species known to be found on Makira below 500m, I did not detect the Makira Boobook or the Makira Moorhen. Consequently, I suggest that thorough assessments of the populations and ecology of endemic species should be a research priority in order to appropriately design management plans and conserve such species. This is an urgent requirement as many of the endemic species detected in this study appear to be specialists of intact forest, and specialists in pristine habitats have a higher risk of extinct (Ricklefs and Bermingham 2002). Thus, an important management step will be to conserve the extent and quality of remaining intact forest. Currently the Solomon Islands has ineffective environmental legislation and policy (UNDP 2006). Existing environmental regulations focus on protecting areas above 400m or steep sided slopes (>30°), which are mainly focussed on limiting commercial logging (UN-REDD 2013) in areas that are not commercially viable anyway. Environmental protection measures in the Solomon Islands should be more appropriately aligned with biodiversity value, to better represent naturally occurring habitats from the coast to higher elevations. This approach has also been recommended for other areas in the wider Pacific region (e.g. Mallari *et al.* 2011). Ensuring the protection of intact forest, particularly in areas below 400m will be important, as these areas are currently unprotected and often under the greatest pressure from logging and plantation development (Buchanan *et al.* 2008).

The geographic isolation of the Pacific region has led to its unique biodiversity, but also generates heightened threats, such as from invasive species, as well as presenting a major barrier to the spread of knowledge and development of support networks. In order to address the rapid environmental change many Pacific Islands are facing, there

needs to be improved networking across the region, not only to move environmental sustainability higher up national agendas but also to create consolidated approaches for conservation.

Chapter 7: Overall synthesis

Overall thesis synthesis and critical discussion

7.0 Chapter Overview

Thus far in the thesis, through establishing an appropriate measure of household wealth, I have identified poorer households to be more dependent on natural resources for their cash income, whereas wealthier households are reliant on cash crops. In addition, I have found cash crop areas, particularly where planted as a monoculture to be a poor habitat for many species of bird, particularly endemic species. In this Chapter I bring together the knowledge I have established on the relationship between poverty and biodiversity from preceding Chapters and introduce some new data gathered through semi-structured interviews, to discuss the land use transition from subsistence agriculture to cash crops occurring in Kahua in the context of the theory of social-ecological traps. I identify common features of social-ecological traps from a review of the relevant literature, in order to be able to assess and discuss the extent to which a social-ecological trap may be occurring in Kahua. The overarching theory of social-ecological traps provides a broader systems view with which to explain the relationship between poverty and biodiversity and how it may be influenced, within the context of land use change. Through re-contextualising the land use change occurring in Kahua as a linked social-ecological process, I aim to provide insights into how it can be addressed from both conservation and development perspectives. This Chapter provides a first attempt at determining if a social-ecological trap is occurring in Kahua. The data I use to assess whether a trap is occurring or not, is based on static 'snap-shot' data, and does not assess temporal change. In addition, in this Chapter I indicate

how the chapters of this thesis (and so any publications arising from them) form a coherent body of work, the contributions I have made to this work, and finally how the work contributes significantly to the expansion of knowledge.

7.1 Introduction

There is an explicit assumption that conserving biodiversity (or reducing the rate of biodiversity loss) can help in efforts to tackle global poverty and enhance human wellbeing (Roe *et al.* 2014). But efforts are currently undermined by a lack of empirical evidence for this relationship. Thus, understanding the relationship between biodiversity and poverty is essential to inform conservation and development strategies that protect elements of biodiversity and the services it provides that are of particular importance to the wellbeing of the poor. Therefore, to address this knowledge gap and also to inform conservation and development initiatives, the over-arching aim of this thesis was to assess the relationship between poverty and biodiversity, within the context of land-use change, and explore the ways in which this relationship may lead to a social-ecological trap. To achieve this aim, and as stated in Chapter 1, the objectives of this thesis were to:

1. Clarify key terms, including poverty and biodiversity
2. Establish an appropriate measure of poverty, through evaluating whether a participatory approach can be used to gain locally-appropriate measure of household wealth inequality
3. Assess how household wealth relates to use of biodiversity through consumption and sources of income
4. Assess how biodiversity is changing with increasing intensity of land use
5. Use a broader systems view and overarching theory of social-ecological traps to explain the relationship between poverty and biodiversity and how it is changing within the context of land use change.

To achieve my aim, I used Kahua, a remote area of the Solomon Islands as a case study to explore this relationship. The Solomon Islands provide an ideal case-study as they contain high levels of biodiversity, are the poorest country in the Pacific region (OECD 2012), and are undergoing rapid social and environmental change (Aswani and Sabetian 2010; Lauer *et al.* 2013). Please refer to Chapter 3 for further details on Kahua, and the Solomon Island context.

Objective 1 was achieved in Chapter 2, which provided a review of the ecological, social and interdisciplinary literature to clarify key terms and identify reasons for the lack of success in conservation and poverty alleviation initiatives. Objective 2 was achieved in Chapter 4 which used a participatory approach to gain locally-appropriate measure of household wealth inequality. I found poor households characterised by fewer household members of a working age and fewer male members, in addition to ownership of fewer assets, which were correlated with lower amount of land tenure. Objective 3 was achieved in Chapter 5 which assessed how household wealth relates to use of biodiversity through consumption and sources of income. I found household wealth influenced the use of natural resources, with poorer households having a higher dependence on natural resources both in terms of daily household food consumption and sources of income. Wealthier households on the other hand, were less dependent on natural resources, with both a lower consumption of wild foods and the majority of their cash income earned from cash crops. Objective 4 was achieved in Chapter 6, where I assessed how biodiversity is changing with increasing intensity of land use and found that cash crops, particularly where planted as a monoculture, appeared to be a poor habitat for many species of birds, with fewer endemic species and changes in composition of ecological traits in this habitat.

Overall, it would appear that livelihoods of poorer households are more dependent on natural resources, but opportunities for them to diversify to other activities are constrained through reduced access to land and labour assets. This is further compounded by the fact that current economic development interventions in Kahua promote cocoa to increase income generation, which appears to be benefitting wealthier households with the subsequent environmental degradation resulting in potentially negative impacts for the wider community. Thus, in this final Chapter, I bring together findings from previous chapters, in addition to new information gathered through semi-structured interviews, to discuss the land use transition from subsistence agriculture to cash crops occurring in Kahua in the context of the theory of social-ecological traps (Objective 5). I use a broader systems view and overarching theory of social-ecological traps to explain the relationship between poverty and biodiversity and how it is changing within the context of land use change. Through re-contextualising the land use change occurring in Kahua as a linked social-ecological process, I aim to provide insights into how it can be addressed from both conservation and development perspectives.

7.1.1 How is the relationship between poverty and biodiversity changing within the context of land use change?

Land use transitions can be major drivers of deforestation and other types of habitat degradation and are one of the primary causes of global environmental change (Lambin *et al.* 2001; Geist and Lambin 2002). In recent years, some of the most intensive land use and land cover changes have occurred in tropical areas (Alves 2002; Arroyo-Mora *et al.* 2005; Liu *et al.* 2006). These regions have become increasingly globalized within the last two decades, and as a result have experienced additional changes including increased commercialization and trans-border trade, and new economic, social and political alignments (Mertz *et al.* 2005). Rural communities

have had to adapt to such changes, often through changing local land use strategies, such as agroforestry systems, cash cropping, mechanization of farming, and crop-livestock integration (Byron 1995; Cairns and Garrity 1999; Mertz *et al.* 2005). However, there are increasing concerns over the sustainability of land use practices and the implications for the rural poor who are most vulnerable to the impacts of land degradation (Wadley 2002; Millennium Ecosystem Assessment 2005).

One of the major land use changes that has occurred over different timescales and in different global historical periods is the shift from traditional or subsistence farming to more intensive agricultural systems (Mertz *et al.* 2012). Traditional subsistence agriculture provides the main or supplementary source of livelihood for millions of people globally (Mertz *et al.* 2009; Ziegler *et al.* 2012). Such traditional agriculture at the tropical-forest-agriculture frontier has long been considered a major driver of deforestation and biodiversity loss (O'Brien 2002). Political and economic pressures may encourage or enforce changes from such systems to more permanent and intensive agriculture practices, including the emergence of new markets for cash crops and various forms of conservation and development policies (Van Vliet *et al.* 2012; Adams *et al.* 2013).

Increasing intensity of land use brings both benefits and costs, highlighting the complexities and inequalities associated with land use transitions. For example, the transition has generally increased household incomes, but has also led to negative effects on the social and human capital of local communities, often exacerbating inequalities and increasing conflicts (Dressler and Pulhin 2010; Rist *et al.* 2010). In addition, the transition has been linked to social benefits, such as improvements in health, education and social networking (Cochran 2008; Fox *et al.* 2008; Dressler and Pulhin 2010), but is almost always associated with negative environmental impacts,

including permanent decrease in forest cover, substantial biodiversity losses, increases in weed pressure, decreases in soil fertility, accelerated erosion, declines in stream water quality, and potential reductions in sequestered carbon (Barlow *et al.* 2007; Fox *et al.* 2008; Xu *et al.* 2009; Rist *et al.* 2010; Van Vliet *et al.* 2012).

With the projected increases in population and consumption, particularly in developing regions (Cincotta *et al.* 2000; Sodhi *et al.* 2013), agricultural change, expansion and intensification is likely to increase. As such changes occur, there is potential for some systems to move towards increasingly impoverished states that are both highly undesirable from welfare and environmental perspectives and that may be very difficult to reverse. These situations, which involve damaging positive feedbacks exacerbating social and environmental problems, have been called social-ecological traps (Cinner 2011; Steneck *et al.* 2011; Kittinger *et al.* 2013). Understanding the factors and processes that create and maintain traps is fundamental to finding effective pathways for escape from persistent poverty, inequality (Maru *et al.* 2012) and ecosystem degradation. Identifying traps early is crucial to avoid the significant impacts on human well-being that may be difficult or expensive to reverse.

7.1.2 An overview of traps from the literature

Traps have been discussed widely in the literature, and have been used as a concept to explain social and ecological processes that produce environmental degradation and livelihood impoverishment (Boonstra and de Boer 2014). The Tragedy of the Commons (Hardin 1968) is a classical metaphor used to capture the type of social dynamics leading to environmental degradation. However, case studies have shown that there can be different outcomes in common-pool resource management: in some situations tragedies are unavoidable, but in others people find ways (intentionally or unintentionally) to overcome the dilemma (Ostrom 1990). Consequently, the Tragedy of

the Commons metaphor and its underlying assumptions have been heavily criticized for its disregard of context (Ostrom 1990; Boonstra and de Boer 2014). Contemporary studies try to identify the conditions that hamper or enable effective common property management (Agrawal 2003), and a leading metaphor in these studies is the idea of "social–ecological traps".

Traps are broadly used in both social and ecological realms to refer to a situation where individuals or communities start in a direction or relationship that later proves to be undesirable or lethal, and from which there is no easy escape (Platt 1973; Costanza 1987; Hoff and Sen 2005). Social-ecological traps are defined as 'situations when feedbacks between social and ecological systems lead toward an undesirable state that may be difficult or impossible to reverse' (Cinner 2011). The critical distinction between social-ecological traps and other types of traps discussed in the literature is the reinforcing nature of feedbacks between social processes and ecological dynamics, which may amplify the initial conditions causing the problems (Kittinger *et al.* 2013). These factors may then interact and reduce the resilience of the social-ecological system. Resilience is the capacity of the social-ecological system to absorb recurrent stochastic events (e.g. natural disasters, economic or political turbulence) and to continue to function without changing fundamentally (e.g. Walker & Meyers 2004). Changes within the system are often gradual (e.g. habitat loss, accumulation of pollutants, emergent markets, changes in values; Folke *et al.* 2004; Biggs *et al.* 2009) and tend to go unnoticed until a threshold is reached, upon which there is a shift in the dominant feedbacks that leads to sudden and long lasting changes in the system structure and function, termed a regime shift (Folke *et al.* 2004). Once a regime shift occurs it may be difficult or impossible to reverse (Scheffer 2009). Avoiding a regime shift and ensuring the continued provision of ecosystem services requires promoting

resilience of the ecological system in the face of change (Gunderson and Holling 2001).

Biodiversity is assumed to underpin ecological resilience in the face of change (Peterson *et al*. 1998), with increased resilience in more diverse biological communities (Cardinale *et al.* 2003). Functional diversity is thought to determine ecosystem functioning through the diversity and values of traits of species present, with species loss often linked to declines in ecosystem services (Hooper *et al.* 2005; Cadotte *et al.* 2011; Hooper *et al.* 2012). Certain species may have irreplaceable roles in ecosystems (Wenny and Levey 1998; Cordeiro and Howe 2003; Şekercioğlu *et al.* 2004) and consequently, changes in their numbers and distributions may lead to disruption of vital ecosystem processes and delivery of services (Redford 1992). These changes may push ecosystems close to thresholds or into alternate states with lower capacity to generate ecosystem services for society, which may lead to a social-ecological trap.

The resilience of the ecological system is also influenced by human activity, with people adapting their behaviour in response to ecological change (Folke *et al.* 2010). Thus, social and ecological resilience are linked, particularly for social groups or communities that are directly dependent on ecological resources for their livelihoods (Adger 2000). There is considerable heterogeneity within the social system, with different social groups who have varying environmental priorities, natural resources claims and power relations (Leach *et al.* 1999). This leads to differences in how people are able to respond to changes within the social-ecological system, with certain social groups better able to adapt and generate benefits from changes than others (Coulthard 2008). This may lead to increasing inequalities, and the perpetuation of a social-ecological trap for those who are not able to adapt to change.

There are various factors within the social system that can impact its resilience or stability, and thus lead to a social-ecological trap, including: 1) inequalities: a certain threshold of resource endowment may be required for households to adapt to change, generate benefits, or even escape poverty (Kelly and Adger 2000; Barrett and Swallow 2006); 2) livelihoods dependent on a narrow range of resources: this increases the variance of income and decreases the stability of the social system through susceptibility to fluctuations in markets and/or the occurrence of extremes events such as droughts, flood or pests and diseases (Adger 2000); 3) Formal and informal institutions (defined broadly as socialised behaviour, as well as formal structures of governance and law): are closely linked to a stable social system when they are inclusive with a high degree of trust (Harriss and De Renzio 1997) and have been shown to be critical for effective natural resource governance (Berkes 1989; Ostrom 1990; Agrawal and Gibson 1999). However, institutions are subject to external pressures and shocks associated with both political and economic change (Adger 2000). For example, the introduction of high financial returns from resources can cause changes in the social organisation for managing the resources that can increase inequalities and social conflict, further undermining support for collective management (Kelly 1996; Adger 2000).

Emerging research on social-ecological traps suggests there are several key feedbacks between social and ecological domains that drive the system toward a trap, including interactions between poverty and resource use, missing or weak institutions, and overharvesting of natural resources, often associated with the use of specific technologies (Cinner 2011; Lindenmayer *et al.* 2011; Steneck *et al.* 2011).

Interactions between poverty and resource use can lead to a social-ecological trap through a reinforcing feedback between increasing resource degradation and

inequalities. For example, in Kenya, poor fishers with fewer livelihood options were the least likely to be able to exit the fishery when resources decline severely (Cinner *et al.* 2009a). This is consistent with a broad body of literature on poverty traps, which demonstrate a series of constraints on the poor, such as limited assets, lack of access to cash or credit, which prevents them from accessing alternative, higher risk and higher income livelihood strategies (Adato *et al.* 2006; Carter and Barrett 2006; Cinner 2011). As a result they choose livelihood strategies with low or short-term returns, and become trapped in a stable or increasing poverty (Barrett *et al.* 2006).

Missing, weak or ineffective institutions can also contribute to a social-ecological trap. Across many Least Developed Countries there is a widespread presence of customary practices and taboos that regulate the use, access and transfer of resources (Cinner and Aswani 2007). Such informal, customary institutions have been shown to be effective at improving or maintaining ecological conditions, either through controls of specific technologies or closures of certain areas to harvesting (Cinner and Aswani 2007). However, these customary systems appear to break down as societies experience certain types of social and economic change (Ruddle 1993), including increasing population (Agrawal and Goyal 2001), and improved market access (Cinner *et al.* 2007).

Such missing, weak or ineffective institutions can often lead to unsustainable harvesting of natural resources (Bennett *et al.* 2007). Overharvesting threatens the sustainability of the resource base, and in certain areas has led to severe species loss (Fa *et al.* 2002) and exacerbation of poverty (Davies 2002). People tend to try to maintain yields as the resource base is depleted, which often leads to the use of technologies that are more damaging to the environment, such as the use of smaller

sized nets in fisheries (Cinner 2009). This leads to a cycle of ecological destruction and increasing poverty (Walker *et al.* 2009), perpetuating a social-ecological trap (Fig. 7.1).

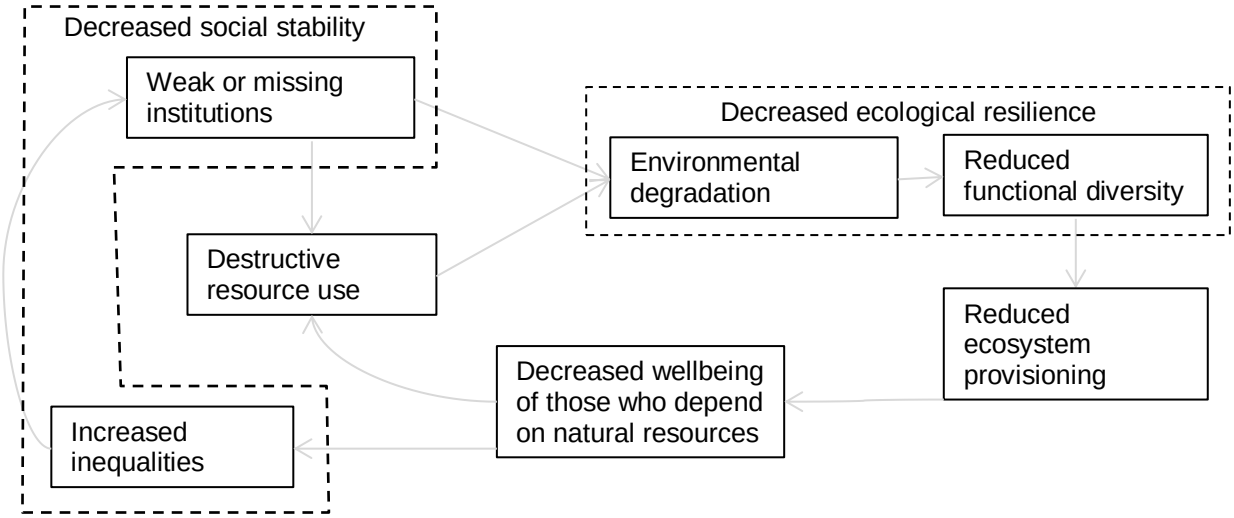

Figure 7.1: Model of a social-ecological trap. Weak, missing or ineffective institutions within a system can drive destructive resource use, which leads to environmental degradation. This reinforces ecological feedbacks that drive the system towards a less desirable state with reduced ecosystem functioning and decreased wellbeing of those who depend on ecosystem services, which then leads to further destructive resource use as people try to maintain yields to support their livelihoods. The cycle continues, with increasing inequalities, with those who do not depend on natural resources for their livelihoods able to sustain their income and consumption regardless of increasing environmental degradation.

In conclusion, common features of a social-ecological trap are: (1) decreased ecological resilience, (2) decreased social stability, (3) interactions between poverty and resource use, and (4) overharvesting of natural resources, often associated with the use of specific technologies. In the following sections I will use these four features, which have already been covered in detail in the preceding chapters, to assess and discuss the extent to which a social-ecological trap is occurring in Kahua.

7.1.3 Overview of Kahua case study in relation to traps

The case-study for this thesis is the Kahua region of Makira (see Chapter 3 for details). Previous research in Kahua has identified population growth, increasing interaction

with the cash economy and desire for material prosperity to be key drivers of change (Fazey *et al.* 2011; Kenter *et al.* 2011). Local people have reported a diverse number of outcomes relating to these changes, including declines in availability of natural resources, declines in food crop yields, increases in land disputes and declines in social cohesion (Fazey *et al.* 2011; Kenter *et al.* 2011). Importantly, major land use transitions are occurring in the region, driven by development policies and aid that promote economic development through the promotion of cash crops. As such, cash crops (primarily cocoa) have been planted in the low-lying, fertile flood plains, with subsequent displacement of food gardens to more marginal lands, such as steep-sided slopes. There are signs of environmental degradation, with declines in primary productivity detected at a landscape scale (Garonna *et al.* 2009). The proliferation of cash crops within communities has also had social repercussions. Positive outcomes from the increased levels of money within communities include the increased ability of people to afford essential services such as school fees and transport. However, increasing money in communities has also been linked to negative social behaviours, including an increase in jealousies, a behaviour described by Kahua people as having a significant impact on social relations (Fazey *et al.* 2011), and an increase in alcohol consumption, which frequently takes priority over school fees and medical expenses (Foale, 2008).

Overall, the broad trajectory of change appears to be taking Kahua people towards more impoverished social and environmental conditions. However, the extent to which the change is creating a social-ecological trap remains unclear. In particular, data have been lacking on the extent of environmental degradation and subsequent changes in the social system. For example, it seems likely that the transition to cash crops will benefit some households, but increase pressure on others and on the environment leading to environmental degradation and greater inequalities. Livelihoods dependent

on cash crops will be maintained despite environmental degradation because income (and accumulation of assets) becomes largely independent from environmental quality, possibly even until very large degradation occurs. However, those who are not able to capitalise on the environmental change may get caught in a trap, with declining resources and few opportunities to escape.

7.2 Summary of data collection:

This thesis was intentionally interdisciplinary and has been built on both social and ecological data. Data were collected using a participatory approach where possible to facilitate the exploration of local knowledge and collect locally-relevant data. I used a variety of data collection methods, including: focus groups to identify local indicators of wealth (Chapter 4); household interviews to collect information on household use of natural resources (Chapter 5); and avian line transects to determine how birds respond to changes in land use intensity (Chapter 6). In addition, I regularly held participatory workshops for all communities in Kahua. These workshops provided an opportunity for me to not only feedback any findings and validate results, but also enabled me to involve the wider community in the study. This approach required the investment of considerable time and resources, and while it may not have been strictly essential for much of my data collection, it was crucial in enabling me to continue to work effectively with local communities and access their land. Moreover, it was central to helping local communities foster learning about local flora and fauna, endemic species and the importance of conservation on Makira.

The following paragraphs describe the data collection for each of the four features of social-ecological traps in turn, in order to assess and discuss the degree to which the land use transition from subsistence agriculture to cash crops in the Solomon Islands

may be creating a social-ecological trap. Where data come from other chapters in this thesis, I reference it accordingly.

1) Are there signs of decreased ecological resilience?

As detailed in Chapter 6, I used changes in biodiversity as a proxy for ecological resilience because high levels of biodiversity appear to enhance ecological resilience (Isbell *et al.* 2011; Admiraal *et al.* 2013), and species loss is closely linked to loss of ecological function (Hooper *et al.* 2012). Specifically I focussed on terrestrial birds, because birds are taxonomically and functionally diverse and provide key ecosystem services (Padoa-Schioppa *et al.* 2006; Whelan *et al.* 2008; Robledano *et al.* 2010). Birds have also been shown to be excellent indicators of agricultural intensification (Gregory *et al.* 2005; Everard 2008). I explore signs of decreased ecological resilience over different land use types. There is no temporal ecological data for Kahua, and therefore I substitute space for time.

I sampled the avifauna across five major land use types, recording all birds by sight or sound along 500 m line transects across a gradient of land use intensity, from intact forest, secondary forest, subsistence food gardens, mixed cocoa, and monoculture cocoa plantations (for a full description of different habitats refer to Chapter 6, and see Appendix A9). I established two transects per habitat type, and repeated each transect a minimum of six times during the survey period. For each habitat the number of species, number of endemics, species composition, and ecological traits (microhabitat selection: water, aerial, canopy, canopy/understory, understory, or terrestrial; and foraging guild: fruit and other vegetative material, nectarivore, omnivore, insectivore or vertebrates), were calculated and compared. These data and their analyses are detailed in Chapter 6.

2) Are there signs of decreased social stability?

To examine the stability of the social system I assess institutional change and presence of inequalities, which have been demonstrated as proxy indicators of social stability (see Adger 2000). I chose to focus on informal institutions because they have been shown to be important for management of common pool resources but that can break down under certain types of change (Cinner and Aswani 2007). To examine any changes within informal institutions, I gathered qualitative data through semi-structured interviews with 38 individuals from 11 villages (Appendix A11). Respondents ranged in age from 27 to 92 years. Respondents were selected opportunistically, depending on who was available in the village. A total of 32 men and 6 women were interviewed. This imbalance in gender sampling arose as women tended to be away from the village in their gardens during the morning, and busy with household chores in the afternoon. I led all interviews in Kahuan through the use of a translator who was familiar with the questions and research project, unless the respondent expressed a preference for communicating in Pijin. I started all interviews by asking the respondent about the village where they had grown up, and moved on to any changes they had noticed within their villages, how people behave, and local livelihoods (see Appendix A11). Questions were open-ended and offered respondents the opportunity to expand and offer information on what they considered important. Interviews were kept as informal and conversational as possible. Interviews lasted an average of 42 (± 24) minutes. These data were also collected between January and July 2012 and add further detail to the information presented in previous chapters of this thesis.

I also recorded asset ownership at the household level because inequalities in assets can affect how households are able to respond to change. Asset ownership is a better measure of long-term, structural poverty (rather than short-term, transitory poverty), and thus provides a better indication of a trap (Carter and Barrett 2006). As discussed

in Chapter 4, twelve focus groups were conducted in six villages (separately for male and female; see Figure 4.1 for map of villages). Participants were asked to identify items or characteristics that changed across three different wealth categories (poor, average and wealthy), and could be considered indicators of wealth. How these indicators changed across the wealth categories was then discussed. The total list of indicators from all focus groups was then presented and discussed at a workshop, where a shortlist of five asset indicators was identified: number of pigs, number of chickens, number of cocoa trees, number of coconut trees and number of gardens. A household survey was used to collect information on ownership of these five asset indicators at the household level. I applied a Principal Components Analysis to the asset data, and the factor scores from the first principal component (which explained 36% of the variation) were used as the wealth score for each household. This was a continuous variable (-2.07–6.49), with a higher value associated with a higher household wealth. We divided households into three wealth categories using the 40th percentile (i.e. the lowest 40% as "poor", the next 40% as "average" and the top 20% as "wealthy", following Filmer and Pritchett 2001) which is a standard approach, in line with World Bank suggestions for poverty analysis (Vyas and Kumaranayake 2006). I then explored household characteristics of these three wealth groups (please refer to Chapter 4 for further details on the methodology).

3) Are there interactions between poverty and resource use?

As detailed in Chapter 5, I assessed household wealth in relation to the use of natural resources in terms of income and consumption. Data on both daily household dietary intake and household budgets were collected through interviews using the 24-hour recall technique (Bingham 1987). A total of 74 households were visited once every two weeks (total of 776 surveys). All foods and beverages consumed during the previous 24 hours, together with their mode of procurement (foraged/hunted, purchased or

received as a gift), and any items purchased, sold or given away by the household were also recorded. The 24hr-recall data was used to construct household consumption (defined as the market value equivalent of all foods consumed). Records of cash income and household expenditures were recorded separately. I explored the relative values of wild foods in terms of household consumption and proportion of household income. These data and their analyses are detailed in Chapter 5.

4) Are resources being overharvested? Is there changed use of harvesting techniques?
I gathered qualitative data through semi-structured interviews (following the methods outlined in point 2 above), with 38 individuals from 11 villages Open-ended questions focussed on any changes respondents had perceived in the abundance of natural resources, numbers of people hunting, and harvesting practices (see Appendix A11). These data were also collected between January and July 2012 and additional detail to support the information presented can be found in the previous chapters of this thesis.

7.3 Summary of results:

In this Chapter I bring together a range of social and ecological data from the preceding Chapters of this thesis to assess the relationship between poverty and biodiversity, in the context of land use change. In Chapter 4, I found poor households in Kahua were characterised by fewer household members of a working age and fewer male members, in addition to ownership of fewer assets, which were correlated with lower amount of land tenure. In Chapter 5, I found household wealth influenced the use of natural resources, with poorer households having a higher dependence on natural resources both in terms of daily household food consumption and sources of income. Wealthier households on the other hand, were less dependent on natural resources, with both a lower consumption of wild foods and the majority of their cash income earned from cash crops. In addition, in Chapter 6, I found that cash crops, particularly

where planted as a monoculture, appeared to be a poor habitat for many species of birds, with fewer endemic species and changes in composition of ecological traits in this habitat.

7.3.1 Considering the key findings from this thesis in relation to traps

To assess how the relationship between poverty and biodiversity, within the context of land-use change in the Solomon Islands, and how this relationship may lead to a social-ecological trap, I use the data from previous chapters and discuss how these comprise the four features of social-ecological traps (Table 7.1).

Table 7.1: The features of social-ecological trap used to explore the extent to which it is occurring and the corresponding evidence from Kahua

Feature of social-ecological trap	Evidence for occurrence in Kahua
Reduced ecological resilience (impaired functions and processes, reduced functional diversity (Elmqvist *et al.* 2003)	Changes in distribution of avifauna ecological traits across increasing intensity of land use: Most intensive land use cash crops (monoculture cocoa) had least endemic avifauna (Chapter 6)
Decreased social stability (inequalities, resource dependency, missing or weak institutions) (Adger 2000)	Breakdown in informal institutions (semi-structured interviews, this Chapter) Signs of narrow resource dependency among all households, with wealthier households dependent on cash crops, and subject to market fluctuations, and poorer households dependent on natural resource (mainly pandanus and hard woods) (Chapter 5) The poor have fewer assets, access to land and less income (Chapter 4 and 5)
Interactions between poverty and resource use (Cinner *et al.* 2009a)	Poor households depend on natural resources for income (pandanus and hard woods) Wealthier households rely on cash crops Poorer households consume most wild foods (Chapter 5)
Overharvesting of natural resources (use of specific technologies)	Reported use of more destructive techniques Reported declines of many resources Increased opportunity of harvest (semi-structured interviews, this Chapter)

1) Are there signs of decreased ecological resilience?

As detailed in Chapter 6, I found differences in avifauna with increasing intensity of land use, with monoculture cocoa areas appearing to be a poor habitat for birds. I found significant differences in species assemblages, with endemic avian species appearing particularly susceptible to increasing intensity of land use. I also saw significant differences in the relative distribution of ecological traits displayed by birds between habitats, including a proportional increase in nectarivores and decrease in insectivores in the most disturbed habitat, monoculture cocoa. Secondary forest and garden areas had high species richness, but many endemic species appeared to be forest specialists. The most intensive land use, monoculture cocoa, appeared to have the largest negative impact on endemic fauna and avian functional diversity.

2) Are there signs of decreased social stability?

82 % of respondents reported changes in informal institutions, including decreasing respect for elders and a breakdown in kastom. Kastom is derived from the English word “custom” and is a generic term that refers to practices that are locality and tribe specific (see Table 3.1). Items that were mentioned as signs of change in kastom, were less sharing of food within communities, less hunting of wild pig, young people being influenced by Western fashions, including women wearing shorts. All respondents who were older than 50 years of age also talked of taboo forest sites that were no longer respected. Taboo sites generally refer to sacred areas of the forest that are culturally important, particularly prior to the arrival of Christianity. Taboo sites are an example of an informal institution and can play an important role in natural resource management (Colding and Folke 2001). The reported signs of a breakdown in kastom and customary management of natural resources reflects a breakdown in informal institutions, which could be considered a sign of decreased social stability.

As presented in Chapter 4, I found inequalities in household ownership of assets (number of: pigs, chickens, coconut trees, cocoa trees, and gardens). The poor owned fewer of these assets, because these assets require land, collectively less ownership of these assets can be considered to be correlated with a lower amount of land ownership at the household level. Exploration of household characteristics of the three wealth groups also revealed inequalities in available labour across households, as poor households had fewer members of a working age and fewer male members.

3) Are there interactions between poverty and resource use?

As detailed in Chapter 5, I found a significant association between household wealth and sources of income. Wealthy households earned the majority of their income from cash crops (88%), followed by 62% for the average-wealth households and 23% for poor households. In contrast to average and wealthy households, poor households derive the majority of their income (53%) from natural resources, including pandanus and hard woods. I also found wealth to be correlated with wild food consumption, with the average-wealth households consuming more wild foods than other wealth categories.

4) Overharvesting of natural resources (use of specific technologies)

Semi-structured interviews revealed that all 38 respondents reported rapid environmental change with many resources important to local livelihoods reported to have declined. Respondents cited a noticeable change in the environment occurring approximately 35 (±13) years ago. All respondents reported substantial declines in the availability of rattan palms and hard wood timber species (which are used for house building). Older respondents reported that some hard wood species could no longer be found, and all respondents described an increase in travel time to collect rattan palm, which in some areas has increased to a full day's walk. With regard to terrestrial

animals, the Northern cuscus (*Phalanger orientalis*) was reported to have declined the most, followed by the wild pig (*Sus scrofa*). 66 % of respondents mentioned an increase in the use of more destructive harvesting techniques. Examples were given of wasteful resource use, for example, cutting trees for house building, but abandoning the cut sticks when better ones were found; more destructive methods, such as cutting down trees (which tend to be large) to harvest flying foxes or possums, rather than climbing the trees and checking the hollows; and use of more destructive technologies, including chainsaws and axes.

7.4 Main conclusions

In this thesis I have demonstrated evidence of key features of social-ecological traps within Kahua, including changed ecological functions across different land uses, a break down in informal institutions, and interactions between poverty and natural resource use. Importantly, I have found signs of inequalities among households, which appear to affect how they are able to respond to change and indicate that a social-ecological trap may be occurring for poorer households. In addition, it would appear that the social-ecological trap is partly driven by the livelihoods of wealthier households which are associated with higher environmental degradation.

As detailed in Chapter 6, I found intact forest areas had the highest proportion of endemic and restricted-range avifauna, a finding which is consistent with other studies that suggest replacing forests with more intensive land uses results in a shift toward bird communities with less specialized traits and a reduction in the diversity for some key functional groups (Şekercioğlu 2012; Luck *et al.* 2013). The change in feeding guilds I observed suggests bottom-up restrictions such as habitat and resource limitations (e.g. food availability). These changes indicate a change in the way the ecosystem is functioning. For example, the reduced number of insectivores in the

cocoa areas could have implications for the spread and control of pests and diseases, which in turn may impact human livelihoods through reduced yields. Exclusions of insectivorous birds from a variety of trees and shrubs has been found to significantly increase insect pests and consequent plant damage (Şekercioğlu *et al.* 2004). These changes also need to be viewed in the context of the island system, where the loss of species that fill key functional roles may inordinately disrupt vital ecosystem services and processes, making them less resilient to other environmental changes (Hooper *et al.* 2005). In addition, many islands have already experienced an extinction filter (arrival of humans, introduction of rats etc.), and therefore future changes or extinctions may push these communities toward functional collapse (Boyer and Jetz 2014). While there are detectable changes in avifauna responses to land use change, data should be sought from other taxa to improve the robustness of this study. In addition, the extent to which this corresponds to the system approaching a threshold remains unknown. Long-term monitoring of change and responses to perturbations within the system can provide insights to proximity to such critical thresholds (Scheffer *et al.* 2012).

As discussed under 7.3.1, local people reported declines in informal institutions, such as kastom, which could indicate reduced social stability. This corresponds with the reported widespread decline of customary tenure across the Pacific (Young *et al.* 2006), which has been linked to the displacement of local ecological knowledge by other forms of knowledge as local social-ecological systems shift from a subsistence livelihood to a market dominated regime (Lauer *et al.* 2013). There was a high degree of resource dependency across all wealth categories, with wealthier households dependent on cash crops and poorer households reliant on natural resources for their income. Thus, the livelihoods of all households in Kahua are, to varying degrees, vulnerable to sudden shocks, such as pests and diseases, or market fluctuations.

The occurrence of inequalities relating to asset ownership and income within communities are also signs of reduced social stability (see Fig. 7.2). In addition, the inequalities in household assets, which all require land, can collectively be considered to be correlated to inequalities in land tenure. My findings offer support to a study by Coomes *et al.* (2011) who found initial landholding conditions in shifting cultivation systems to have long-term effects on household welfare. For example, households may be unable to accumulate sufficient land, equipment, or livestock, to adopt more productive agricultural systems and thereby move out of poverty (Dercon 1998; Barrett *et al.* 2006; Barrett and Swallow 2006; Carter and Barrett 2006). Thus, household's initial assets would appear to strongly influence households' livelihood strategies and consequently their future welfare and whether they become trapped in poverty (Figure 7.2, and see Coomes *et al.* 2011). How inequalities are changing

As discussed in Chapter 5, I found reinforcing relationships between livelihoods of the poor and natural resources, with the poor earning the majority of their income from natural resources, a relationship that corresponds with other studies (e.g. Fisher 2004; Shackleton and Shackleton 2006; Fu *et al.* 2009). In contrast, I found wealthier households to be somewhat less dependent on natural resources, both in terms of their sources of income, which were heavily based on cash crops, and in terms of consumption, where they were found to consume the least amount of wild foods.

The lower involvement of poor households in cash cropping suggests that the poor have less access to such income sources, either through a lack of initial land holding assets and/or other assets such as education or labour. This is consistent with the view that the harvest of natural resources offers the best return for labour input where access to capital, land or livestock is limited (Shackleton and Shackleton 2006). Consequently, wealthy households who appear to have more land available to plant

both food gardens and cash crops, and enough household labour to be able to adequately achieve both of these dual strategies, are able to take advantage of the opportunities presented by land use change. This gives rise to another reinforcing feedback, through the degraded environmental conditions associated with the cash crop areas and livelihoods of the wealthy and increased opportunity costs of the livelihoods of the poor. Although the livelihoods of all communities in Kahua are dependent on ecosystem services to some extent, as the wealthy increase their cash income they become increasingly detached from the environment as they are able to pay for services (e.g. food, water, fuel), whereas the poor are tied to livelihoods linked to ever diminishing natural resources and become trapped in a downward cycle of poverty and environmental degradation with little opportunity to escape.

Local people reported declines in many natural resources and an increase in destructive harvesting techniques and practices. This combination of factors suggests that many species that are important to local livelihoods are being over-harvested. Species which are prone to over-exploitation tend to have a restricted distribution, high habitat specificity and low reproductive rates (Sodhi *et al.* 2009). Worryingly, these are features of several species utilised in Kahua. For example, 75% of climbing *Pandanus* species found in the Solomon Islands are endemic (Pacific Horizon Consultancy Group 2008).

7.4.1 Wider implications of a social-ecological trap for conservation and development

The consequence of variation within communities has serious implications for both conservation and development initiatives. Communities and their responses to environmental change are often viewed as homogenous units and treated as such by conservation and development initiatives (Agrawal and Gibson 1999).

However, increasing research is demonstrating the importance of recognising the heterogeneity of communities and their responses to change (Coulthard 2008), and the need to collect disaggregated data for monitoring impacts of conservation initiatives (Daw *et al.* 2011). I suggest this approach should also be extended to social-ecological traps.

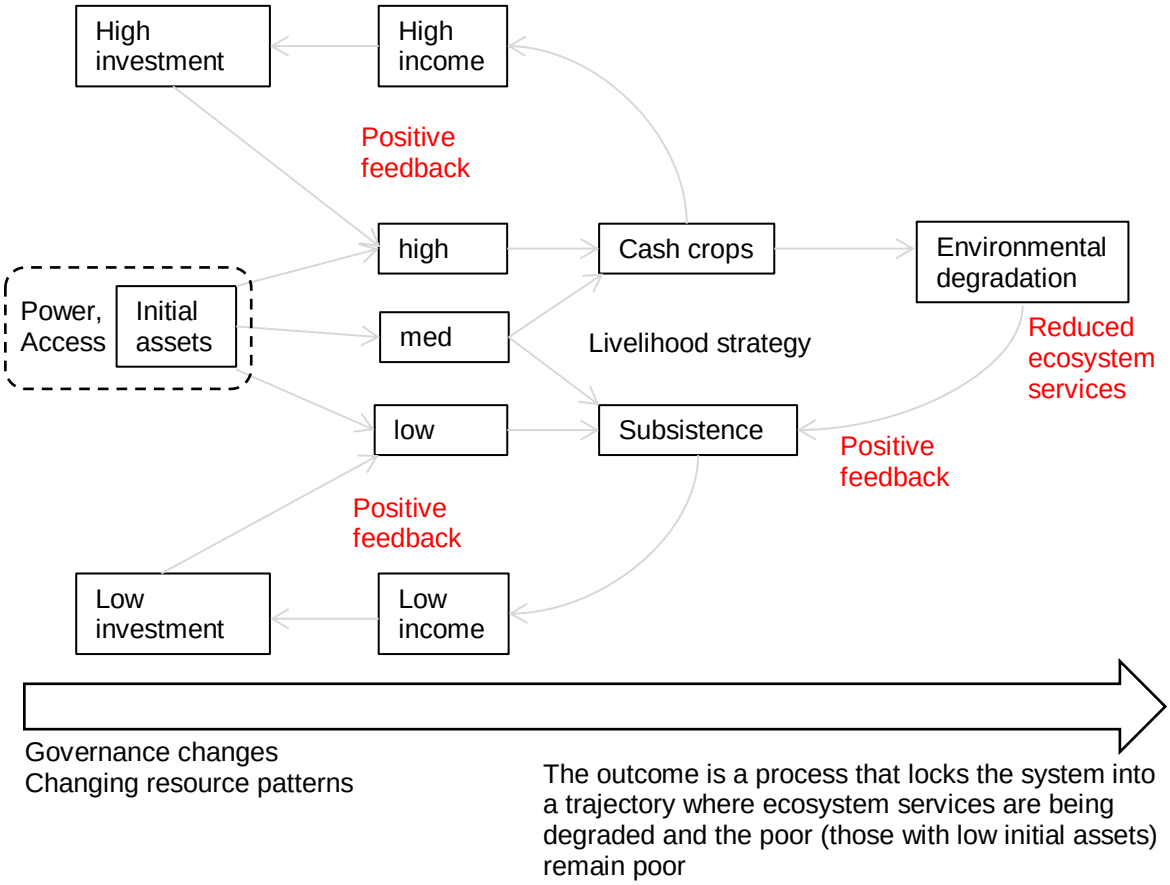

Figure 7.2: Model of how a social-ecological trap could develop in Kahua. Initial household assets influence the livelihood strategy they are able to pursue which impacts the return and thus investment. The outcome is a process that locks the system into a trajectory where ecosystem services are being degraded and the poor (those with low initial assets) remain poor.

The promotion of cash crops by development agencies in Kahua appears to be a driver of ecological degradation and increasing inequalities, resulting in a social-ecological

trap for poorer households. This has arisen partly because development projects tend to be focussed on income generation, so while aggregate incomes may increase, growing inequalities remain hidden. In addition, household's initial assets are often overlooked, but are crucial in determining how households are able to adapt to change. Consequently, the benefits from income-focussed development projects tend to exacerbate inequalities in land distribution (McSweeney and Coomes 2011), leading to trans-generational social-ecological traps (Coomes *et al.* 2011; McSweeney and Coomes 2011). A focus on income-generation also overlooks potentially significant environmental degradation. Thus, development needs to go beyond conventional income generation, capacity building, and even governance reform to include support for local peoples' struggles for land and resource access (McSweeney and Coomes 2011), with particular emphasis on supporting those who are most marginalised. In addition, a long-term view of sustainability should be considered. Development programmes cannot rely on the assumption that resource conditions will improve with socioeconomic development and ignore potentially serious regime shifts (McClanahan *et al.* 2002). Livelihoods that depend on a narrow range of resources are vulnerable to external shocks, such as market fluctuations and pests and disease. To overcome this, future income-generation programs in rural communities should support a variety of activities with differing system requirements, such as timing of labour demands and spatial requirements (Eriksen and O'Brien 2007; Brooks *et al.* 2009). While these may be more costly compared to more focused programs (Prowse and Scott 2008), such initiatives will enhance subsistence security (McSweeney and Coomes 2011). Furthermore, governments and donor agencies should make sustainability a cornerstone of development programs so that projects that aim to contribute to poverty alleviation as part of natural resource management do not inadvertently result in increasing contributions to larger-scale threats to ecological degradation (Dietz *et al.*

2007; Cinner *et al.* 2009b). Activities which adopt a long-term approach and consider inequalities will help prevent inadvertently creating a social-ecological trap.

At present, there are no conservation initiatives in Kahua, but the high level of endemism threatened by land use change and importance of natural resources to local livelihoods suggests conservation action is required. The Solomon Islands currently has ineffective legislation and policy framework (UNDP 2006) resulting in weak environmental protection. Environmental regulations currently focus on protecting areas above 400m or steep sided slopes (>30 °), which are mainly focussed on limiting commercial logging (UN-REDD 2013), in areas which are not commercially viable anyway. The findings in this thesis suggest that the extent and quality of intact forest should be maintained. In addition, I propose that habitat protection in the Solomon Islands should be aligned more appropriately with biodiversity value and represent more fully naturally occurring habitats. In Kahua, the monoculture cocoa is currently spatially constrained, but should their development continue on the current trajectory, negative repercussions for biodiversity and ecosystem function seem inevitable. Management actions will need to focus at scales relevant to key social and ecosystem processes and drivers (Steneck 2009). In Kahua, this could include protection of the remaining forest, improving ecological connectivity throughout the landscape, discouraging planting cocoa as a monoculture, and propagation of important plant species for livelihoods of the poor. In addition, as much biodiversity loss, both in Kahua and other systems, is driven by powerful global processes linked to the world economy, conservation needs to start to engage with development processes at this level to move sustainability to the centre of the development agenda. Ignoring these wider economic and social dimensions of global change and development consigns conservation to a marginal position on the side-lines of debate about global development and sustainability (Adams 2013).

7.4.2 Limitations of this thesis and further study

Kahua presents an ideal case study to examine the relationship between biodiversity and poverty in the context of land use change, because local people have a high dependence on the natural environment and the region is undergoing rapid environmental and social change, particularly due to an increasing interaction with a market economy. Thus in many ways, Kahua can be considered a microcosm for studying the pressure and complexity of globalization on remote island systems. On the other hand, the Kahua region represents a small region of the Solomon Islands, which is a notoriously diverse cultural area. Also, it is important to bear in mind that the main findings in this thesis derive from a snapshot study (main data collection period was six months) from one region of the Solomon Islands, which inevitably limits the study's representativeness. The scope of this thesis was inevitably a compromise between maximising data quality (repeating transects, sample size) and feasibility (travel time between study sites, contingency for lack of access, e.g. during inclement weather). Thus, I hope that future studies will further build on the insights gained during this research and complement its shortcomings.

While the approach I used in this thesis to measure wealth inequality goes beyond basic income measures, it is still restricted to material dimensions of poverty. This approach does not reflect less tangible dimensions of poverty, such as social capital, access to resources or power relations, which are likely to be important factors that influence a household's ability to exploit natural resources and pursue different livelihood strategies. Thus, further research should focus on attempting to measure such dimensions, in an approach that remains applicable and replicable for conservation and poverty alleviation initiatives.

One aspect of this thesis focussed on household use of natural resources, which have been shown to be an important resource for the rural poor in other studies (de Merode *et al.* 2004; Schulte-Herbrüggen *et al.* 2013). However, the collection of additional data on natural resources important to the livelihoods of the poor other than wild foods, such as firewood, building materials and traditional medicines, would provide additional information on the relationship between poverty and biodiversity. In this study, I also attempted to assess the use of building materials and medicinal plants, but I found these resources had no equivalent market value and were difficult to identify, and in the case of medicinal plants were used in small quantities. To overcome these shortcomings and better assess the importance of biodiversity to local livelihoods and the poor, involving specialists in data collection, such as botanists, could help with the identification of species, and other measures of resource valuation could be adopted (e.g. participatory valuation).

In addition, although birds have been shown to be a good indicator of biodiversity and the impacts of land use change, the extent to which they are congruent with other taxa in the Solomon Islands should be explored. I originally attempted to survey a wider selection of taxa, including birds, bats, frogs and butterflies. However, the methods I used to survey butterflies (baited traps), which had been found to work in other tropical forest environments (e.g. Hughes *et al.* 1998), failed to capture any butterflies. While frogs are abundant and diverse throughout the Solomon Islands, on Makira there are only two species of frog: an endemic species (San Cristobel frog, *Rana krefftii*) and the invasive cane toad (*Rhinella marina*). Initial frog surveys only recorded cane toads and were consequently terminated. Acoustic monitoring of bats surveys proved more successful (see publication in Appendix A1), but were complicated through the lack of a reference call library (calls already identified to species level), which are needed to accurately identify species, in addition to constant heavy rain, which could damage the

sensitive microphone. Birds are probably the most well studied taxa in the Solomon Islands, yet there is still a lack of detailed ecological data, and as such ecological processes that promote and maintain avian diversity are unknown (Kratter *et al.* 2001). The Kahua region represents a small area of Makira, and therefore broader inferences about biodiversity should be conservative. However, the Kahua region is one of the only areas on Makira not to have experienced commercial logging and as such the Kahua region may represent a more intact avifauna than other areas of Makira. In order to be able to determine this, further studies should include avifauna surveys across Makira Island.

This thesis provides a first attempt at determining if a social-ecological trap is occurring in Kahua. While there is evidence for many of the key features occurring in Kahua, a more robust determination of whether a social-ecological trap is occurring or not, could be achieved with the collection of additional data. Specifically, further data on changes within the ecological system would enable a better determination of how a trap is forming and whether the system is approaching a critical threshold. While I found changes in ecological traits with increasing intensity of land use, the extent to which this is resulting in changes in ecosystem functioning on a landscape scale is unknown. More research is needed to identify keystone species (those that undertake crucial support functions) within the ecosystem, such as seed dispersal, pollination, or pest regulation (Hougner *et al.* 2006). In addition, ecological resilience also refers to how a system is able to return to stability following a perturbation (Holling 1973). Thus, data on a set of variables which are associated with self-organization would improve this study, and could include information on the size of the resource system, the productivity of the system, and resource unit mobility (see Ostrom 2009). A critical data need is comprehensive time series data for major social and ecological states (Millennium Ecosystem Assessment 2005). Such information can be generated from

the paleoenvironmental records, such as lake sediments, ice cores and tree rings (see Dearing *et al.* 2012b). Such multi-decadal perspectives are not only useful for providing context, but may represent the true timescale within which the contemporary system operates, helping to observe trajectories and identify the existence of thresholds(Dearing *et al.* 2010; Dearing *et al.* 2012a). In addition, to truly determine if species that are important to local livelihoods are over-exploited, further data are needed, including information on species abundance, life history traits and offtake (Milner-Gulland and Akçakaya 2001). Furthermore, while there were reports of the use of more destructive harvesting techniques, I do not know who the users are of such techniques. For example, in the Kahua system, it could be assumed to be the poor as they are more dependent on natural resources for their income, but likewise it could be wealthier households who have more capital to afford more expensive technologies, such as chainsaws. Understanding these intricacies is important to further identify drivers of change within the system.

In this thesis I found that household's starting conditions appear critical for their future wellbeing and that certain trajectories of change may magnify inequalities within communities, leading to a social-ecological trap for poorer households. Social-ecological traps are a useful concept for re-contextualising land use change as a linked social-ecological process to provide new insights into how it can be addressed from both conservation and development perspectives. However, as with predicting regime shifts, identifying traps is inherently difficult. Emerging research has identified key features of social-ecological traps, but more research is required to translate this concept into management actions, such as how many of these features create a trap, whether there are different levels or depths to a trap, and how these can be identified and measured over time. In addition, a particular focus for further research should be on the role of informal institutions and how these maintain inequalities and/or create

barriers to adaption at the community level. Identifying less tangible dimensions of poverty, such as social capital and power, may also inform causes of poverty and how inequalities are maintained. Once barriers have been identified, ways to address them can then be developed.

Tropical oceanic islands provide an ideal system for assessing the relationship between biodiversity and poverty because they are generally biodiversity hot spots (Mittermeier *et al.* 1998) and tend to have stronger linkages between ecosystem services and people local people (Millennium Ecosystem Assessment 2005). However, the tightly coupled terrestrial and marine ecosystems of islands means that the impacts of natural or anthropogenic changes are much more immediately visible than for larger continental systems (Brookfield 1990). Island systems also experience more specific challenges and vulnerabilities, compared to larger continental areas, arising from the interplay of socioeconomic and environmental factors, such as small populations and economies, weak public- and private-sector institutional capacities and remoteness. For example, invasive species have caused serious ecological and economic damage and high social costs across many island systems (e.g. Lowe *et al.* 2000; Courchamp *et al.* 2003). In this respect, the findings of this thesis may be more pronounced than would be expected in larger continental areas.

7.4.3 Implications and recommendations for conservation and development from research in this thesis

Despite the limitations discussed above, this thesis has provided findings that are relevant to both conservation and development, in the Solomon Islands and more generally. Firstly, this thesis has demonstrated the value of a participatory approach for measuring wealth inequality, which is suitable for areas with limited engagement with a cash economy, complex family and tribal ties, and an absence of basic infrastructure. I

found the results from this approach to be well correlated with household expenditure and that it provided insights into poverty, in a manner that was appropriate to the local context. This participatory approach could improve the effectiveness of community-based conservation through identifying the poor and exploring local poverty in other regions. The knowledge gained from this exercise could then be used to develop appropriate strategies to address the underlying causes of poverty, alongside providing a baseline measure from which to monitor impacts of any intervention.

Secondly, in recognition of the contributions that natural resources make to livelihoods of poorer households in Kahua, efforts should be taken to improve the sustainability and security of this relationship. In Kahua, the proportion of income earned from natural resources by poorer households was higher than found in other studies (Chapter 5, section 5.4.2). This may reflect that the market economy was introduced far later than in other developing regions (Furusawa and Ohtsuka 2006), and so there remains a distinct lack of options for income generation, but a growing need for cash income as people become more integrated into the market economy. As with other areas around the world, the collection of natural resources is an activity that is generally available to all households, but one that is more likely to be exploited by poorer households with limited land resources and other assets (e.g. Fisher 2004; Shackleton and Shackleton 2006; Fu *et al.* 2009). The use of natural resources clearly plays a vital role in supporting daily livelihoods of poorer households in Kahua. To improve the sustainability and security of this livelihood resource, efforts could include assessing the feasibility of cultivation and sustainable management of species identified by local communities as important to livelihoods (e.g. *Pandanus* sp.). However, how the use of such natural resources could assist these households to accumulate assets or improve their standard of living remains less clear. Other studies have concluded that the use of

natural resources are more central to preventing the deepening of poverty, rather than lifting people out of poverty (Shackleton *et al.* 2007).

Thirdly, this thesis highlighted the critical role that inequalities can play in constraining access to different livelihood opportunities, which has wider implications for conservation and development initiatives to ensure they do not exacerbate inequalities. For example, the lower involvement of poor households in cash cropping suggests that the poor have less access to such income sources, either through a lack of initial land holding assets and/or other assets such as education or labour. Thus, this thesis indicates there is a need for development initiatives to go beyond conventional income generation projects, to include support for differential land and resource access (McSweeney and Coomes 2011), which will most likely require institutional reform. However, this may be particularly challenging in Kahua and across the wider Melanesian region because of the somewhat unique customary land tenure. This collective ownership of land and natural resources by descent-based groups means that development and conservation initiatives need to operate differently in this region. Customary land tenure has been considered to present an insurmountable barrier to commercial agriculture and foreign investment, as there are multiple claimants to a particular parcel of land, no strong tradition of delegated authority, and no statute of limitations with regards to customary claims (Fukuyama 2008; Haque 2012). Consequently there are added challenges for landowners to convert customary land into alienable property, which both denies potential investors secure tenure, and also inhibits entrepreneurialism by preventing the use of land for collateral to access loans. This may mean that the inequalities arising from land tenure in the Solomon Islands, and across the wider Melanesian region, are more entrenched than in other areas around the world.

Fourthly, the negative impact of monoculture cocoa on avian diversity, particularly endemic species, should be addressed. My findings in relation to the increasing intensity of land use change and a decrease in endemic species are in line with other tropical island studies in São Tomé (de Lima *et al.* 2012), and more generally with an increasing body of literature demonstrating habitat specialists to be particularly affected by environmental disturbance (Colles *et al.* 2009). Specifically, intensive cocoa farming has also been found to have negative impacts on biodiversity in other studies, with declines noted across a wide range of plant and animal taxa (e.g. Lawton *et al.* 1998; Schulze *et al.* 2004). The negative impact on biodiversity associated with increasing monoculture plantations fits into recent literature that has found there to be 'habitat fragmentation threshold', which once crossed eventually leads to a regime shift (Pardini *et al.* 2010). This highlights the need to conserve a reasonable amount of native vegetation cover to maintain high levels of biodiversity and ecological resilience in human-modified landscapes (Pardini *et al.* 2009). Thus, in Kahua the expansion of monoculture cocoa plantations could be expected to negative impact on endemic biodiversity and ultimately ecosystem functioning. Consequently efforts should be made to prevent the expansion of monoculture plantations in Kahua. For example, this could be achieved through the promotion of planting cocoa as a shade crop. This is where cocoa is planted under the shade of native tree fauna, and has been shown to have improved benefits for both crop yields and biodiversity (Steffan-Dewenter *et al.* 2007; Waldron *et al.* 2012). For this to be achieved, small holders will need support in how to mitigate the negative impacts of monoculture cocoa on the environment, such as through increasing the numbers of native shade tree species and planting fruit trees.

7.5 The contributions I have made to this work

To discuss my contributions to this thesis, I break down contributions into three stages: planning, data collection, and analysis and write-up. During the planning stage, I

established the research questions for this thesis. Prior to the official start of my PhD I also travelled to Kahua to meet with local communities and develop my research questions. During this stage, I also applied for additional funds for the field work, and sought the relevant ethical approval and the appropriate research permits. I conducted the data collection phase of this thesis by myself in a remote region of the Solomon Islands, where there was little transport or communication infrastructure. During the first year of my PhD I completed a pilot data collection phase (March-May 2011), which allowed trialling of methods and survey method refinement. The main data collection was then conducted during my second year of PhD (January to July 2012). During this time, I trained five local people in all aspects of data collection, including participatory methods, household surveys and basic survey skills, including use of GPS. These five local people were then employed as research assistants to assist with data collection, particularly the household surveys, translation of semi-structured interviews and community outreach work. All the data collection was overseen, validated and entered by myself. I also carried out and completed all aspects of the analysis and write-up stage of the thesis. My supervisors, Ioan Fazey, Nathalie Pettorelli, Will Cresswell and Guy Cowlishaw, provided advice and guidance on methods and approaches, in addition to reading and commenting on draft versions of this work (see breakdown in Table 7.2)

Table 7.2: Breakdown of author contributions to each chapter in this thesis (TD= Tammy Davies; IF = Ioan Fazey; NP = Nathalie Pettorelli; WC = Will Cresswell; GC = Guy Cowlishaw)

		Percentage contributions to different aspects of each chapter (%)				
Thesis Chapter	Author	Conceptual development	Method design	Data collection	Data analysis	Writing
Chapter 1	TD	100	n/a	n/a	n/a	100
Chapter 2	TD	85	n/a	n/a	n/a	97
	WC	2	n/a	n/a	n/a	1
	IF	3	n/a	n/a	n/a	1
	NP	10	n/a	n/a	n/a	1
Chapter 3	TD	100	n/a	n/a	n/a	100
Chapter 4	TD	90	89	100	96	97
	NP	2	3	0	2	1
	WC	2	3	0	2	1
	IF	6	5	0	0	1
Chapter 5	TD	85	85	100	94	95
	IF	4	2	0	0	1
	NP	3	3	0	2	1
	WC	2	4	0	2	1
	GC	6	5	0	2	2
Chapter 6	TD	80	80	100	86	90
	RC	5	5	0	3	2
	JE	5	5	0	3	2
	IF	1	1	0	0	1
	NP	4	4	0	3	2
	WC	5	5	0	5	3
Chapter 7	TD	92	92	100	100	92
	WC	2	2	0	0	2
	GC	2	2	0	0	2
	NP	2	2	0	0	2
	IF	2	2	0	0	2

7.6 How the work contributes significantly to the expansion of knowledge

This thesis has provided empirical evidence for the relationship between poverty and biodiversity. However, it has also highlighted the complexity of this relationship and the influence of social factors, such as informal institutions, power and cultural relations. Social-ecological systems highlight the importance of addressing biodiversity

conservation and poverty alleviation, not as individual phenomena but rather as complex dynamic systems. Using a social-ecological system lens can provide a useful and holistic framework for assessing this complex relationship, in addition to integrating both natural and social sciences (Janssen and Ostrom 2006; Fisher *et al.* 2014), directly addressing one of the current flaws in poverty and conservation efforts (see Chapter 2).

This thesis has provided insights into biodiversity responses to increasing intensity of land use in a tropical island system. However, an urgent goal is to catalogue species and their significant functional traits in accessible databases to enable field-collected species lists to serve as a key to estimating biodiversity in its fuller meaning (Lyashevska and Farnsworth 2012; Davies *et al.* 2013). Achieving both of these factors will then help to establish a detailed evidence base, from which poverty and conservation initiatives could address and monitor. Further information on ecological traits and processes would then facilitate exploration of how changes in biodiversity may correspond to changes in overall ecosystem functioning, the provision of ecosystem services and the well-being of the poor.

In this thesis, I suggest social-ecological traps as a useful concept for re-contextualising land use change as a linked social-ecological process, to provide new insights into how it can be addressed from both conservation and development perspectives. However, effectively translating the concept of social-ecological traps to appropriate management actions requires further work. This may include identifying how different features contribute to creating traps, the 'depth' of traps, how to identify these different levels and the challenges of escaping from them, and identifying the socially-contingent nature of a trap (i.e., a trap for whom?). As an extension from this, an important avenue for future research is the examination of underlying factors that

influence heterogeneities within communities, including power relations and cultural values, and how these in turn affect the institutional dynamics that mediate human-environment dynamics. Exploration of these factors will require further engagement with the social sciences, including insights and critiques about power and knowledge (Cote and Nightingale 2012). Social-ecological thinking has much potential to inform approaches for sustainable resource management.

Social-ecological traps are a useful way to translate the theory of social-ecological systems into action (Kittinger *et al.* 2013) and the concept has great potential to inform approaches for integrated biodiversity conservation and poverty alleviation, and sustainable resource management in general. However, it is important to bear in mind that biodiversity loss and persistence of poverty are symptoms of far larger development challenges, including overconsumption, consumerism and economic growth along pathways chosen by already wealthy countries (Adams 2013). In this context, biodiversity-based poverty alleviation is just a small part of sustainable development. Yet without approaches that address local-scale biodiversity conservation and poverty alleviation, ecosystems will continue to be degraded and their ability to continue to support poor people will be compromised. Thus, an important step is to establish a solid evidence base both for poverty alleviation and biodiversity conservation. This will not only enable replication and scaling up of successful initiatives, but it will also facilitate the assessment of linkages between biodiversity, ecosystem functioning and services provision, and the well-being of the poor.

7.7 Conclusion

Overall, this thesis highlights the consequences of underlying inequalities and the importance of acknowledging how these can influence the distribution of benefits from development and conservation initiatives. This is an area that needs greater research

and monitoring, for example, the extent to which such inequalities are linked to other factors, such as gender, caste, and religion should be further explored. The time I spent with communities in the Solomon Islands, and the difficulties I witnessed with regard to livelihood security and the rate of social and environmental change, re-affirmed my belief that conservation and development activities need to be urgently reconciled.

Throughout the course of my PhD, I have learnt a wide range of skills, particularly with respect to participatory approaches. The use of which during the course of my PhD, changed the way I think about working at the grass-roots level. These approaches improved my capabilities of working effectively with local communities to gain input and feedback on conservation approaches and activities, and I also saw how these can be used to prioritise and design appropriate conservation interventions. The community-wide participatory workshops that I held provided an excellent learning stage for everyone involved, myself included – and I learnt much about how people were thinking, what they knew about their environment and how they perceived change. This in turn helped to shape my ideas and thinking about what was happening in the Kahua system.

In addition, converting my thesis chapters to papers has provided me with an invaluable experience in communicating information effectively, and the peer-review process has undoubtedly improved my research through the input from relevant experts. Moreover, it has enabled me to become more familiar with this process and develop as an independent researcher. Overall, and despite various challenges, I have thoroughly enjoyed the course of my PhD. I feel I have learnt and developed as a researcher immeasurably, and I am looking forward to the next chapter of my research career.

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Appendix A1: Impacts of forest conversion on bat assemblages

Assessing the impacts of forest conversion on bat assemblages in the Solomon Islands

This appendix will be submitted for publication under the authourship: *Ruzicka, F., Pettorelli, N., Walters, C.L., Lavery, T., and Davies, T.E.*

Co-authour contributions: I was responsible for partly-designing this study, all data collection, establishing research questions and assisting with writing of the paper for publication. Nathalie Pettorelli provided input into the study design. Filip Ruzicka analysed the data and produced a longer version of this manuscript for part fulfilment of MRes in Biodiversity, Evolution and Conservation at University College London (2012). Nathalie Pettorelli and Charlotte Walters provided advice and help when designing the analyses. Tyrone Lavery contributed library calls to assist with the analysis. All co-authours read and commented on all versions of this chapter.

A1.0 Abstract

Despite extensive land-use changes on Paleotropical islands, little is known about how such changes are affecting biodiversity in these hotspot areas. To address this knowledge gap, we characterized bat responses to forest conversion in a biodiverse, human-threatened coastal rainforest habitat on Makira, Solomon Islands. Bats are excellent indicator taxa of habitat disturbance and are promising indicators of broader ecosystem health. We analysed ~200hrs of acoustic recordings from echolocating bats in four land-use types: primary forest, secondary forest, garden and cocoa plantation. We expected cocoa plantations and gardens to make poor habitats for bats; we also

expected bat responses to be guild-specific, such that bat species well-adapted to the forest interior should be rarely detected in degraded habitats compared to “edge and gap” species. Multivariate statistical techniques were used to group similar call profiles and infer taxonomy, providing the first acoustic records for many bat species in Melanesia. Relative activity levels suggested that intermediately modified habitats (garden, secondary forest) are favoured by edge and gap bats, while primary forests remain important for some narrow-space foragers. Conversely cocoa plantations have low activity levels. However, these results only approach statistical significance. We discuss possible methodological and ecological rationales for these results, and argue that lightly modified habitats represent important foraging habitats for bats. Overall, this work constitutes the first detailed exploration of anthropogenic effects on mammalian diversity in the Solomon Islands, with findings of important conservation relevance both locally and globally.

A1.1 Introduction

Tropical rainforests are the planet’s most diverse ecosystem, harbouring more than half of all known species (Myers *et al.*, 2000, Wright, 2005). They are also one of the most threatened habitats, at the frontier of agricultural expansion and increasing human influence (Laurance, 1999, Bradshaw *et al.*, 2009). Anthropogenic land use change is a major driver of the current extinction crisis (de Lima *et al.*, 2012) and with only 9.8% of the entire tropical forest biome within strictly protected areas (Schmitt et al. 2008), there is growing recognition of the importance of understanding biodiversity responses to land use change to inform management decisions. In addition, degraded lands are also often important for sustaining the livelihoods of local people; consequently the future of tropical forest biodiversity depends on landscape-level management of people and their impacts on biodiversity (Harvey and Villalobos, 2007, Perfecto and Vandermeer, 2008, Gardner *et al.*, 2009).

Higher intensity of land use can lead to reduced habitat diversity and subsequently decreased species diversity (Fitzherbert et al. 2008; Maitima et al. 2009; Wretenberg et al. 2010). Conversely, intermediate-levels of disturbance can reduce competitive exclusion of certain species and lead to higher species richness, termed the 'intermediate disturbance hypothesis' (Connell 1978; Huston 1979). Therefore, we would expect forest degradation activities will impact species richness in relation to the degree of disturbance. However, there is limited understanding on the impacts of anthropogenic land use change on biodiversity; this is a particular concern as understanding how species respond to land use change can improve management decision making (Rodriguez 2003). Empirical evidence for the impacts of land use change on biodiversity is particularly sparse for tropical islands; despite islands containing disproportionality high number of endemic species that are simultaneously prone to extinction driven by land use change (Woinarski, 2010, Waltert *et al.*, 2011, de Lima *et al.*, 2012).

To address this shortcoming, we assess the relative biodiversity of a mosaic landscape of four habitats: primary forest, secondary forest, subsistence gardens and cocoa plantations using a case study from Makira, Solomon Islands. The Solomon Islands are a biodiversity hotspot in the South West Pacific, with high levels of endemism and remarkably intact forest cover (Bayliss-Smith *et al.*, 2003). The country is experiencing rapid social and environmental change, with an increasing population (2.6% per annum, UNICEF, 2011) and an economy that is heavily based on extractive industries, particularly logging (Kabutaulaka, 2000, Pauku, 2009). The country is also desperately understudied, with little information on how the current rates of change are impacting its unique biodiversity. Our assessment focuses on echolocating bats (suborder: Microchiroptera). Bats are excellent indicator taxa of habitat disturbance at the community level as they are taxonomically and functionally diverse, often abundant,

global in distribution and provide key ecosystem services (Jones *et al.*, 2009). Bats are also sensitive to human-induced changes to ecosystems (e.g. Estrada *et al.*, 1993, Hayes and Loeb, 2007, Kunz *et al.*, 2007, Vleut *et al.*, 2012), and have been used as ecological indicators of habitat quality (e.g. Wickramasinghe *et al.*, 2003, Kalcounis-Rueppell *et al.*, 2007). Population declines suggest that bats are affected by environmental stressors, and that monitoring of their populations may give insight into the importance of these stressors in a more general context (Jones *et al.*, 2009).

We formulate two hypotheses regarding bat responses to land use change. First, we expect that primary and secondary forests will exhibit similar bat activity levels, as previous studies have shown that bat diversity remains high across forest successional stages (Presley *et al.*, 2008, de la Peña-Cuéllar *et al.*, 2012). This in itself may be because forests have high food availability, high level of habitat heterogeneity and low risk of predation (Estrada *et al.*, 2004). It may also result from high canopy cover and high plant diversity (Gorresen and Willig, 2004, Harvey *et al.*, 2006, Castro-Luna *et al.*, 2007); both characteristics of the Solomon Islands' forested habitats (Katovai *et al.*, 2012). We expect lower activity in cocoa plantations, as monocultures are often bat-poor (Estrada *et al.*, 1993, Harvey and Villalobos, 2007, Fukuda *et al.*, 2009, Phommexay *et al.*, 2011). Because gardens have higher plant diversity than cocoa plantations but more open canopies than forests, we conjecture that gardens will exhibit intermediate activity levels.

Second, we hypothesize that the influence exerted by forest conversion will be guild-specific. Bat species which forage in highly cluttered spaces, or "narrow-space" foragers, emit short-distance echolocation calls and have energy-intensive flight (Norberg and Rayner, 1987, Schnitzler and Kalko, 2001, Kingston *et al.*, 2003). For these morphological and ecological reasons, narrow-space foraging species are

thought to be very dependent on intact forested habitat (Law *et al.*, 1999, Kingston *et al.*, 2003). As a consequence, we expect forest conversion to severely affect narrow-space species. By contrast, species which forage in background cluttered space (near the edges of vegetation, in vegetation gaps, or near ground or water surfaces) known as "edge and gap" foragers (Schnitzler and Kalko, 2001) should be found more commonly in disturbed habitats.

In addition to testing these hypotheses, our study addresses several ancillary research gaps. First, it supplements the scant bat literature, whose examinations of responses to forest conversion have been primarily undertaken in the Neotropics (Struebig *et al.*, 2008), with relatively few acoustic method studies from the Paleotropics (Phommexay *et al.*, 2011). In addition, we provide the first acoustic characterizations of bat species common and/or endemic for a data-deficient region.

A1.2 Methods

A1.2.1 Study area

Field work for this study was focussed in the Kahua region ($162^0$0'−$162^0$15' E, $10^0$25'−$10^0$40' S) of Makira Island (formally San Cristobal) (Fig. 1). Makira has an area of 3191 km² and consists of a narrow coastal plain leading up to undulating hills with steep forested central ridges, with elevations of up to 1200 m (Allen *et al*., 2006). It has a wet tropical climate characterized by high humidity and little annual variation (Fasi *et al.*, 2013) and nearly no seasonality (Danielsen *et al.*, 2010). Makira contains a total of 16 bat species (from 10 families), of which 9 are echolocating species (Flannery, 1995). Kahua has approximately 4500 inhabitants across 42 mostly coastal communities. Local livelihoods in Kahua are heavily based on shifting cultivation for subsistence gardens. Cocoa was first introduced to the Solomon Islands in the early 1960s and is now planted in small-holder plantations throughout Kahua. These land

use practices have created a mosaic of forest, garden and secondary-regrowth habitats, with useful trees, such as fruit and nut trees preserved throughout the landscape (Mertz *et al.*, 2012). There are four main habitat types across the region that can be characterised as follows:

Primary forest: closed canopy (30-45m high), composed of large, hardwood trees, including those suitable for timber (*Pometia pinnata, Vitex cofussus, Pterocarpus indicus, Calophyllum vitiense*), with dense understory vegetation including thickets of smaller trees, rattan palms (*Calamus* spp.), *Stenochlaena* ferns and *Slaginella* mosses. Anthropogenic disturbance is a ubiquitous feature of the forests of the Solomon Islands (Bayliss-Smith *et al.*, 2003) and therefore no forests can be considered "primary" in its truest sense. We therefore use primary forest to refer to the lowland evergreen tropical rainforest with, historical, but presently limited human disturbance.

Secondary forest: no continuous canopy, although crowns can be in close proximity to each other, mainly composed of small fast growing, pioneer species, (including *Macaranga* spp*., Fiscus* spp*.,* and *Hibiscus tiliaceus*) indispersed with larger trees, including Ngali nut (*Canarium indicum*), breadfruit (*Artocarpus altilis*), coconut (*Cocos nucifera*) and sago palm (*Metroxylon salomonense*). Often used intensively.

Garden: open canopy consisting of interspersed low-lying food crops such as yam (*Dioscorea* spp.), taro (*Colocasia esculenta*), sweet potato (*Ipomoea batatas*) and slippery cabbage (*Hibiscus manihot*)*,* as well as various protected or deliberately planted herbaceous and tree species, such as coconut palms, a wide range of banana and plantain cultivars (*Musa cultivars*), breadfruit, sago palm, nut trees (*e.g*. *Canarium* spp., *Barringtonia edulis*, *Inocarpus fagifer)*, edible figs (*Ficus* spp.), betel-nut palm (*Areca catechu*), mango (*Mangifera indica*), and papaya (*Carica papaya*).

Cocoa: smallholder plantations of the cocoa tree (*Theobroma cacao*) which typically grows 4-8m high. Cocoa trees are planted close together resulting in a closed canopy with a clear understory (that is maintained). The cocoa plantation is often interspersed with coconut trees and occasionally lone large trees such as breadfruit (*Atrocarpus altilis*) and Malay apple (*Syzygium malaccense*).

A1.2.2 Acoustic monitoring

Ultrasonic monitoring of bats was conducted from 14th February – 10th July 2012 using the fixed location ultrasonic recorder SM2BAT 384 kHz and omnidirectional SMX-US microphone (Wildlife Acoustics, www.wildlifeacoustics.com). The SM2BAT is a 16-bit full spectrum recorder, which preserves the amplitude and harmonic details of the original bat signal and is more sensitive to detecting bat calls. It was programmed to record continuously from sunset and sunrise, relative to local times. Consistent with recent recommendations (Meyer *et al.*, 2011), recordings were undertaken for a minimum of four nights in each of the habitat types. The detector was attached to a tree at a height of at least 1.5m, with the microphone slightly pointing down to protect it from rain damage. The detector was positioned at multiple sites for each of the four habitat types, (i.e. each sampling night was at a different location), rather than repeated sampling at one site (following Gorresen *et al.*, 2008). To minimise any effect of altitude, all sampling was conducted below 500m. Positioning the SM2BAT was constrained by the rugged terrain and the nature of the land-use mosaic across the study site, but care was taken to position the SM2BAT as widely as possible to increase independence of sampling. To avoid damage to the microphone, the SM2BAT was not positioned during nights of heavy rain.

We visualized all sound recordings on a spectrogram included in BatSound v3.31 software (Pettersson Elektronik, AB, Uppsala, Sweden) (sample rate 384 kHz, FFT-

size 512, Hanning window). Files containing noise were discarded. Any ambiguous calls were excluded from statistical analyses. Among files containing bat calls, we selected only discrete sequences of search-phase calls where fewer than two bats (or bat species) were present in a file, in order to maximize sequence independence (Gannon and Sherwin, 2004). We used SonoBat v3.0 software (Szewczak, Arcata, CA, USA) to quantify a variety of acoustic parameters from each call pulse, utilising the "Sonobatch" feature to circumvent manual bias during the SonoBat selection process, but manually rejected poorly fitted calls to ensure a degree of quality control.

A1.2.3 Statistical Analyses

For all statistical analyses, we selected the highest quality call (defined by a high signal-to-noise ratio, as computed by SonoBat) from each sequence to represent one bat pass. Where bat passes had readily recognizable shapes and/or frequencies they were categorized visually into different call type groups. However, when call characteristics did not clearly indicate categorization, a cluster analysis was used to distinguish between calls. Cluster analysis provides an objective way of grouping ambiguous bat calls because it can be informed by call parameters known to distinguish well between calls (Walters *et al.*, 2012). All ambiguous calls in our dataset belonged to Miniopteridae, we therefore used six parameters recently shown to discriminate well between bat species (Walters *et al.*, 2012; Table A1.1).

Table A1.1: Definitions of call parameters used in cluster analysis (from: www.sonobat.com/SonoBat%20parameters.html)

Call Parameter abbreviation	Full name and definition
Fc	Characteristic Frequency
LowFreq	Lowest call frequency
FreqMaxPwr (FMP)	Frequency of maximum amplitude
HiFreq	Highest frequency of the call
CallDur	Duration of the call
FreqLedge	Frequency the most abrupt transition to the most extended flattest slope section of the body of the call preceding the characteristic frequency. This is referred to as the "ledge" of the call
FreqCtr	Frequency at the centre of the duration of the call
FreqKnee	Frequency at which the initial slope of the call most abruptly transitions to the slope of the body of the call

We ran cluster analyses separately in each habitat because call profiles differed slightly between habitats. To determine the appropriate number of clusters for each habitat we plotted parameters used in clustering against those not used (*e.g.* call duration). This indicated that four clusters were appropriate for most habitats. The ANalysis Of SIMilitude (ANOSIM) test was used to test for any statistical differences between cluster-generated call-type groups. ANOSIM was deemed appropriate for our dataset because our parameters could not be known to meet the assumptions of parametric testing. We evaluated the relative performance of various clustering methods (*e.g*. different number of clusters, different dissimilarity indices, hierarchical versus non-hierarchical clustering) using ANOSIM which identified the complete-linkage hierarchical clustering with Bray-Curtis dissimilarity as the most appropriate clustering method with consistently higher R values than other techniques. All cluster analyses were carried out in R v. 2.15.2 (R Development Core Team, 2012). Within R, we used the *vegan* package to conduct ANOSIM tests with Bray-Curtis dissimilarity. In order to assign clusters to species, we compared our recorded call profiles with previous

studies, as well as reference calls (Anabat and Time-Expanded) obtained from recent study of captured bats on Makira, and confirmed our assignations via previous acoustic surveys (*e.g*. Jones and Corben, 1993, De Oliveira, 1998, Hughes *et al.*, 2011, Robson *et al.*, 2012).

To test our hypotheses regarding bat responses to habitat conversion, we measured activity levels (bat passes per night) for each species. Though commonly used, we emphasize that activity level is only an approximation of true abundance (Walsh *et al.*, 2004). To test for significant differences between habitats, we used non-parametric Mann-Whitney U tests (or the equivalent Kruskal-Wallis tests when comparing between more than two classes). We performed this procedure on combined species data to test our first hypothesis, but separated bats into guilds and species to test our second hypothesis. The lack of information about the biology of Solomon Island bats prevented accurate guild assignation, so we grouped species into guilds from family-level categorizations (Schnitzler and Kalko, 2001). Because of the low total number of bat species found on Makira, we elected not to produce or interpret estimates of true species richness in each habitat, though we note them for information.

A1.3 Results

We recorded a total of 1925 bat passes over 17 days (~190 h) across all four habitats. This translates to 11.23±1.15SD mean recording hours per night (Cocoa=11.98±0.31SD h; Garden=10.44±1.57SD h; Secondary=10.98±1.01SD h; Primary=11.6±1.14SD h). Many calls had characteristic frequencies (Fc) between 35 and 60 kHz, and appeared to separate into four call type groups. These calls had frequency modulated and curving pulses, with constant frequency tails, and sometimes terminated with visible downward sweeps. Through cluster analysis, we were able to consistently show very high differences between the four putative clusters (ANOSIM R

= 0.8565-0.9916, p=0.007-0.12), suggesting that these four clusters represent four discrete call types. This was further confirmed through modifying the number of clusters under ANOSIM testing which produced lower R values. Comparison of call profiles (Table 1) with previous descriptions and reference calls from Makira suggested the highest Fc (~54kHz) call and lowest Fc (~38kHz) call clusters belong to *Miniopterus australis* and *Minioptus propitristis* respectively. The two remaining clusters (Fc~43kHz and Fc~50kHz) were more difficult to assign. Previous descriptions suggest that Fc's between 42 to 47kHz correspond well with *M. schreibersii*; however, reference *M. schreibersii* calls from Makira had Fc's between 48kHz to 53kHz. We therefore grouped the Fc~50kHz cluster with *M. schreibersii*. Reference calls were unavailable for *M. macrocneme,* and this was therefore grouped by default with Fc~43kHz (Fig. A1.2 D-G). Nevertheless, it is possible that Fc~43kHz belongs to *M. schreibersii* and Fc~50kHz to low-frequency *M. australis*, with *M. macrocneme* completely absent.

In addition to Miniopterid calls, we identified three different and distinct call types. One call type is frequency modulated, exhibits short linear pulses with characteristic frequencies at ~55kHz, and terminates with a steep down-sweeping tail dropping by ~10kHz. Another call type is also short and constant frequency, has a characteristic frequency at approximately 118kHz, and generally terminates with a long down-sweeping tail dropping by ~25kHz. Previous descriptions and reference calls from Makira indicate that these calls belong to an Emballonurid (*Mosia nigrescens*) and a Hipposiderid (*Aselliscus tricuspidatus*) respectively. Reference calls were also suggestive that the Fc~69kHz call belongs to *Hipposideros demissus*. It has a constant frequency pulse and a down-sweeping tail (dropping by ~10kHz). These three species exhibit multi-harmonic calls, but recorded call details reflect only the dominant harmonic (Table A1.2; Fig. A1.2 A-C). We also recorded call profiles that correspond

well to *Myotis adversus* (Fig. A1.2 H). This species has not previously been recorded for Makira, but is found on neighbouring islands (Flannery, 1995).

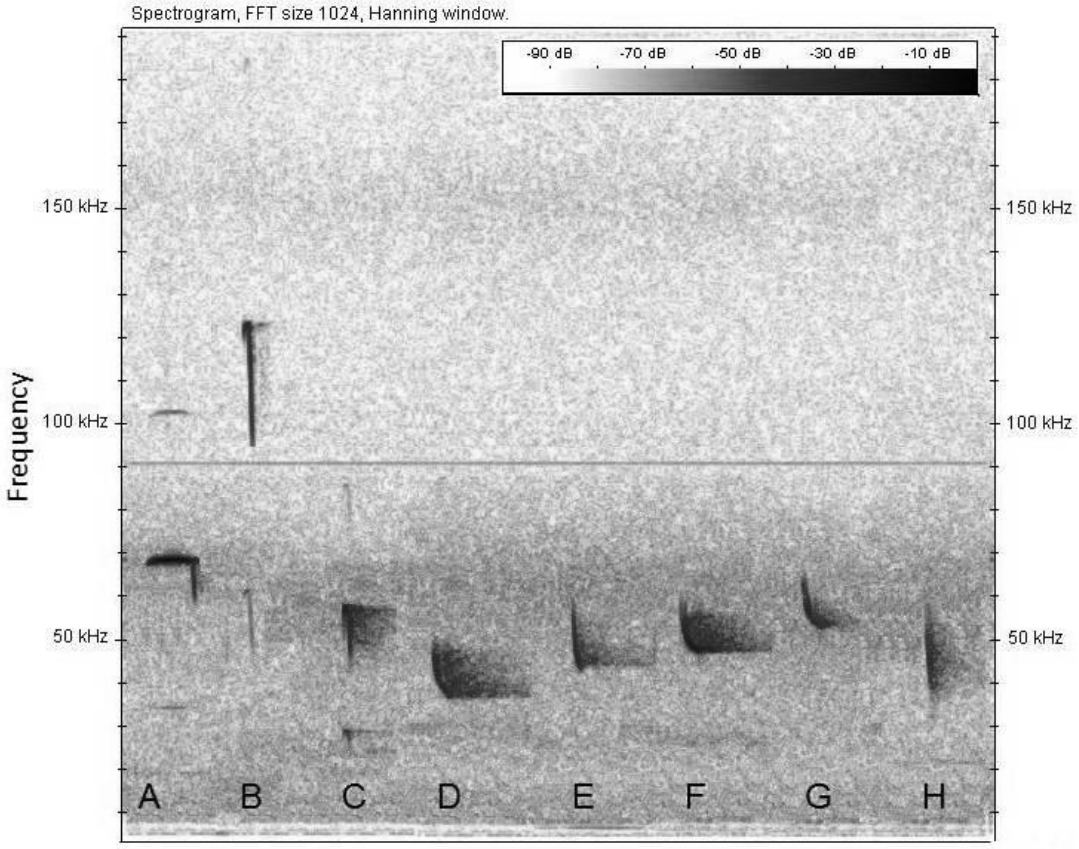

Figure A1.2. Example call profile for each species recorded on Makira: (A) *Hipposideros demissus*; (B) *Aselliscus tricuspidatus*, (C) *Mosia nigrescens*, (D) *Miniopterus propitristis*, (E) *Miniopterus macrocneme*, (F) *Miniopterus schreibersii*, (G) *Miniopterus australis*, (H) *Myotis adversus*.

Table A1.2: Call profiles for all bat species [a]. Values are presented as mean ±SD (min-max)

Species	N	Call duration	Characteristic frequency	Frequency max	Frequency min	Peak energy frequency
Hipposideros demissus	23	13.95 ±3.65 (2.3-18.9)	69.0 ±1.83 (62.2-70.9)	70.1 ±2.2 (67.4-78-7)	64.8 ±4.09 (57.0-69.8)	69.0 ±1.82 (62.3-70.1)
Aselliscus tricuspidatus	153	4.5 ±1.13 (1.628-8.656)	118.6 ±7.29 (94.2-126.4)	122.1 ±2.00 (115.7-128.3)	107.2 ±8.31 (90.4-123.6)	121.1 ±3.93 (94.3-127.1)
Mosia nigrescens	64	3.7 ±1.54 (1.6-10.4)	52.4 ±3.57 (43.2-60.4)	55.3 ±1.91 (52.1-60.4)	47.6 ±3.59 (40.1-58.2)	54.2 ±2.70 (44.7-59.5)
Miniopterus australis	370	3.5 ±1.27 (1.3-8.1)	54.7 ±1.77 (50.7-59.8)	70.7 ±7.15 (59.0-95.9)	54.1 ±1.86 (48.6-59.2)	56.3 ±2.54 (50.7-66.2)
Miniopterus schreibersii*	674	4.7 ±1.78 (1.3-11.4)	49.7 ±1.69 (46.0-55.6)	64.8 ±6.73 (51.5-102.4)	49.1 ±1.71 (45.3-53.0)	50.6 ±2.33 (45.9-57.6)
Miniopterus macrocneme*	309	4.2 ±1.86 (1.1-11.9)	42.8 ±1.81 (38.5-47.2)	54.5 ±4.85 (45.0-71.1)	42 ±1.76 (37.4-46.5)	44.1 ±1.42 (39.7-49.8)
Miniopterus propitristis	332	5.6 ±1.97 (1.5-12.5)	38.2 ±1.30 (34.9-42.6)	51.8 ±5.19 (38.3-70.4)	37.5 ±1.20 (34.6-41.0)	41.5 ±2.47 (35.5-47.2)
Myotis adversus	4	2.8 ±0.81 (2.19-4.01)	43.1 ±10.81 (33.7-57.4)	66.6 ±11.61 (53.15-79.8)	38.9 ±8.29 (29.9-46.3)	48.1 ±6.44 (44.2-57.8)

[a] ** species with uncertain assignation*

A1.3.1 Habitat responses

Miniopteridae, Emballonuridae and Hipposideridae species were found in all habitats, except primary forest, where two *Miniopterus* species (*propitristis* and *macrocneme*) and *H. demissus* were not detected. Using mean bat passes per night as a measure of activity level, we found that most (87.5%) bat passes across all habitats were from the Miniopteridae family. *M. schreibersii* was the most commonly recorded species (35.0%), followed by *M. australis* (19.2%), *M. propitristis* (17.2%) and *M. macrocneme* (16.1%). 7.9% of calls were from *A. tricuspidatus*, 3.3 percent from *M. nigrescens* and 1.2 percent from *H. demissus* (Table A1.3; Fig. A1.3).

Table A1.3: Activity levels recorded for all species (data is displayed as mean bat passes per night (median) ± SD). Percentage of total composition for each species is also noted for all habitats. All estimates are corrected for sampling effort.

			Activity levels				
Family	**Guild**	**Species**	All habitats	Cocoa	Garden	Secondary	Primary
			N = 17	N = 4	N = 4	N = 5	N = 4
All			113.24 (44) ±95.56	41.7 (22) ±50.91	246.5 (149.1) ±271.93	143.2 (74.6) ±204.49	31.7 (39.2) ±18.88
Hipposideridae	Narrowspace	Hipposideros demissus	1.35 (0) ±1.39 (1.2%)	1.2 (0.5) ±1.90	0.5 (0.5) ±0.58	3.3 (0) ±4.61	0
		Aselliscus tricuspidatus	9.00 (3) ±5.90 (7.9%)	1.2 (0.5) ±1.90	7.5 (2.7) ±10.94	5.3 (3.1) ±6.69	22.8 (27.1) ±16.88
Eballonurid		Mosia nigrescens	3.76 (0) ±4.50 (3.3%)	1.9 (0) ±4.00	8.1 (1.1) ±13.69	4.1 (0) ±7.38	1.5 (0.48) ±2.38
		Miniopterus australis	21.74 (4) ±22.40 (19.2%)	3.1 (2.8) ±2.88	40.4 (27.4) ±43.41	40.3 (8.18) ±67.98	2.4 (1.9) ±3.00
		Miniopterus schreibersii	39.64 (11) ±42.52 (35.0%)	7.3 (6.6) ±6.31	78.3 (38.7) ±98.20	67.7 ±117.09	5.1 ±10.50
Miniopteridae	Edge and gap	Miniopterus macrocneme	18.17 (1) ±20.95 (16.1%)	15.9 (1.4) ±32.03	49.0 (20.4) ±67.11	12.1 (7.2) ±13.30	0

Miniopterus propitristis	19.53 (3) ±26.63 (17.2%)	11.3 (1.8) ±21.38	62.7 (16.7) ±92.78	10.4 (5.1) ±12.38	0

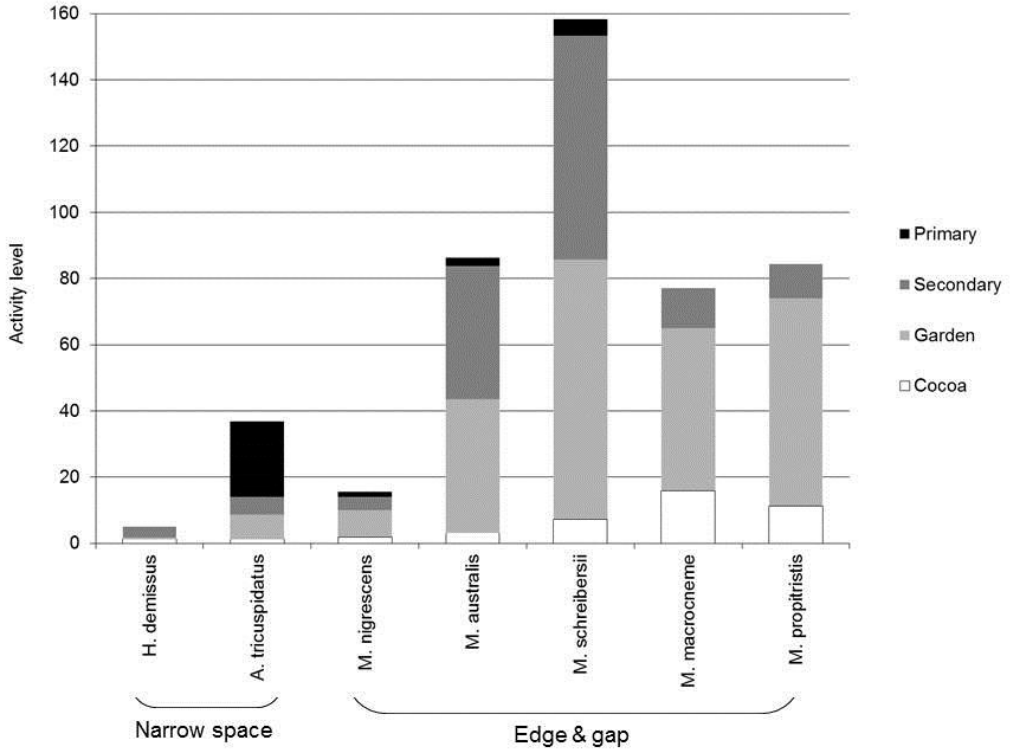

Figure A1.3: Activity levels (mean bat passes per night) for each species, including relative activities in each habitat. Species are also marked by foraging guild.

We recorded very high activity levels in gardens (~246 bat passes (BP)/night), intermediate levels in secondary forest (~143 BP/night), and low levels in both cocoa plantations (~42 BP/night) and primary forest (~32 BP/night) (Fig. 3 and 4). However, there was a great deal of variation between sampling nights within a given habitat, as attested by high standard deviations. Mean bat passes per night were higher than

median bat passes per night in all disturbed habitats, suggesting an important effect of few, extreme sampling nights on mean activity levels (Fig. A1.4; Table A1.3). Tests for differences in overall bat activity between all four habitats produced non-significant results ($H = 3.04 < H_{Crit} = 7.26$, p=0.38), as did tests for differences between cocoa plantations versus secondary and garden ($U_A = 11 < U_{Crit} = 4$, p=0.14). Differences between garden and secondary forest, versus cocoa plantations and primary forest, were non-significant at p=0.05, but significant at p=0.1 ($U_A = 10$, z = 1.4, p=0.08).

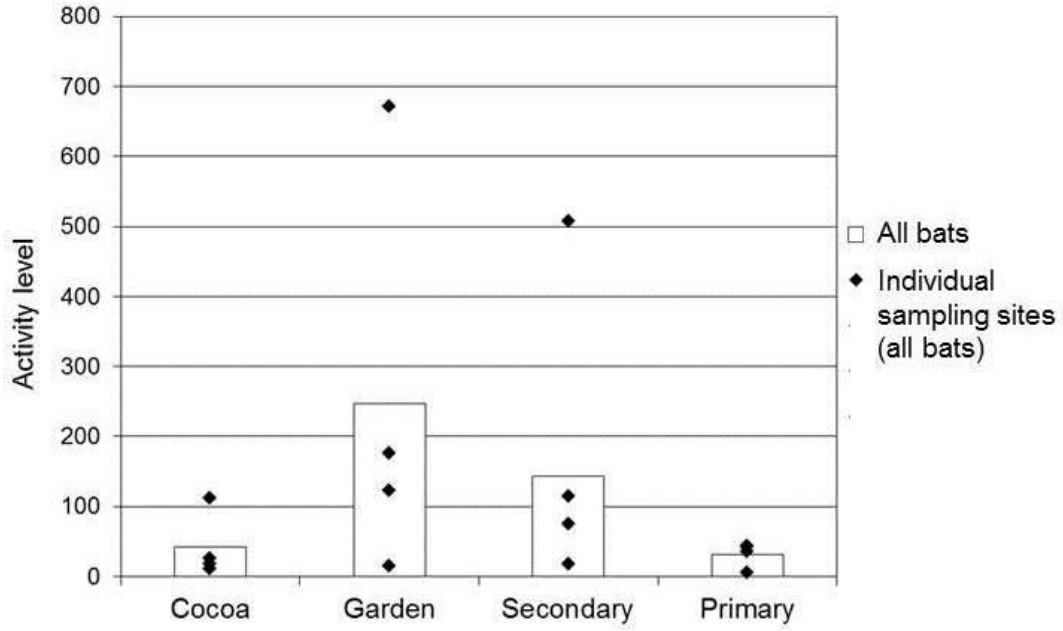

Figure A1.4. Activity levels (mean bat passes per night) for all bats in each habitat type. Points indicate individual sampling points.

When species were split into guilds, trends for edge and gap (E&G) foragers mirrored trends for all bats, with high activity levels in moderately disturbed habitats (~238 BP/night and ~134 BP/night in garden and secondary forest respectively) and low levels cocoa plantations (~39 BP/night) and primary forest (8.96 BP/night). Narrow-space (NS) foragers, on the other hand, exhibited a gradated response to habitat

conversion, favouring primary forest the most and cocoa plantations least (primary forest: ~23BP/night; secondary forest: ~9BP/night; garden: ~8BP/night; cocoa: ~2BP/night) (Fig. A1.3). However, differences for either guild between all four habitats (E&G: $H = 5.2 < H_{Crit} = 7.26$, p=0.16; NS: $H = 3.51 < H_{Crit} = 7.26$, p=0.32), or between moderately disturbed habitats and cocoa (E&G: $U_A = 11 > U_{Crit} = 4$, p=0.14; NS: $U_A = 12 > U_{Crit} = 4$, p=0.18), were non-significant. This conclusion held true at the species level, with the exception of *M. propitristis* ($H = 6.8 < H_{Crit} = 7.26$; p=0.08). For narrow-space foragers, differences between moderately disturbed habitats versus cocoa plantations and primary forest were not significant ($U_A = 38.5$, z = -0.19, p=0.42). For edge and gap foragers, differences were significant ($U_A = 17$, z = 1.78, p=0.04), with four species of Miniopteridae ($U_A = 15.5\text{-}21.5$, z = 1.35-1.92, p=0.03-0.09) accounting for most of this difference.

A1.4 Discussion

Using acoustic monitoring, we were able to detect all but two known species of echolocating bat from Makira. We confidently characterized Makira endemic *H. demissus* for the first time, as well as common Melanesian species *M. propitristis.* Comparisons of total bat activity showed that activity levels differ markedly between sites in a given habitat, but not significantly between habitats, and we therefore reject our initial hypothesis that total activity levels decline in response to anthropogenic modification. Mean activity levels for all species in moderately disturbed habitats were three to seven times higher than in either cocoa or primary forest, with edge and gap foraging species favouring disturbed habitats. Therefore modified tropical habitats in a mosaic landscape have conservation value.

Acoustic recordings have increasingly been used as a means of monitoring bats and other mammals for over a decade (Blumstein *et al.*, 2011). A strength of acoustic

methods is that they capture a more complete and less biased sample of a given area's acoustic biodiversity than traditional capture methods (MacSwiney *et al.*, 2008). In our recordings, all but two species of echolocating bat known to Makira were identified (Flannery, 1995). This included *M. nigrescens,* which has not been caught on Makira via capture methods (Tyrone Lavery, pers. obs.), as well as previously unrecorded *Myotis adversus*. The only species caught by trapping methods which remained undetected in our data was *Hipposideros cervinus*. Acoustic methods therefore produced high inventory completeness, confirming their appropriateness for detecting many species. We note that they may not be wholly appropriate for Hipposiderids, which were rarely detected in our recordings. One of the weaknesses of acoustic methods is that, where there is no established call library, as is the case for the Solomon Islands, species identification is difficult. We addressed this by performing a cluster analysis informed by known discriminatory parameters (Walters *et al.*, 2012), which is a superior alternative to visual inspection (Williams-Guillén and Perfecto, 2011). This is the first study to characterize the acoustic profile for Makira-endemic *H. demissus* and *M. propitristis.* These species were shown to have similar profiles to other species of their family (Robson *et al.*, 2012). *M. macrocneme* and *M. schreibersii* were also characterized, but additional reference calls should be obtained to confidently confirm call profiles.

Mean activities indicated that bats are more abundant in moderately disturbed habitat, although total bat activity levels between habitats did not vary significantly. We propose three methodological reasons for this: (1) our sample size was too low to detect the signal; (2) activity estimates may be confounded by uneven detectability between habitats and species (Patriquin and Barclay, 2003, Meyer *et al.*, 2011). On Makira, primary forests are extremely dense, cluttered environments and detectability is likely to be poor in this habitat resulting in artificially low activity levels; and (3) bats on

Makira are predominantly cave-roosting (Flannery, 1995) and therefore the proximity to cave roosts (which was unknown) could account for much of the variation in activity levels between sampling night and be causing high inter-habitat variation and non-significant results. Supporting this third point, on nights when activity levels were high, most bat passes were recorded within a two to three hour period after sunset which is when bats typically leave their caves to forage. However, a more straightforward explanation for the non-significant variation in total activity is that echolocating bats use habitats indiscriminately. The response of bats to fragmentation is variable among species (Cosson *et al.* 1999b, Estrada *et al.* 1993, Fenton *et al*. 1992, Schulze *et al.* 2000). Bats show a high dispersal capability among landscape elements including open, disturbed and other man-modified habitats (Bernard & Fenton 2003) and have been found to tolerate habitat fragmentation better than other species (Laurance *et al.*, 2002, Bernard and Fenton, 2003). Indeed, some studies have shown that monocultures can be suitable habitats for some species of bat (Pineda *et al.*, 2005, Castro-Luna *et al.*, 2007, Williams-Guillén and Perfecto, 2011). However, we argue against the idea that habitat use is indiscriminate. From our recordings, activity levels never exceeded 30 per night in cocoa, but only once fell below 30 in gardens. Almost invariably, activity levels for species in cocoa plantations were lower than in either secondary forest or gardens. A study in Mexico also found lower richness and abundance of bats in cocoa plantations than forested habitats (Estrada *et al.*, 1993). Cocoa plantations on Makira are fairly open habitats, which represent and an elevated predation risk for bats, as the lack of refuge sites leaves the bats highly visible to predators, such as hawks and owls (Russo *et al.*, 2007). Additionally, such areas offer a limited diversity of food resources relative to well-preserved forest and agroforestry crop systems (Castro-Luna and Galindo-González, 2012, Garcia-Morales *et al.*, 2013). Monocultures reduce the diversity of various arthropods (Perfecto and Snelling, 1995, Perfecto *et al.*, 1997, Watt *et al.*, 1997) and given the insectivorous diet of most

microchiropterans (Fenton, 1982), monocultures are indeed likely to make poor habitats for Paleotropical bat assemblages (Fukuda *et al.*, 2009, Phommexay *et al.*, 2011). In addition, detectability is likely to be comparable to secondary and garden habitats in this study; consequently low activities probably indicate a true unsuitability of cocoa plantations for bats.

In contrast, moderately disturbed habitats generally had high activity levels compared to cocoa plantations and primary forest. The response was guild and species-specific, with edge and gap foragers, particularly Miniopteridae, exhibiting marked preference for this habitat. The morphology of edge and gap bats enables wider habitat use (Schnitzler and Kalko, 2001), and this finding is mirrored in Neotropical studies, where modified agricultural systems have been shown to support viable populations of bats and other taxa (Harvey and Villalobos, 2007, Medina *et al.*, 2007, Williams-Guillén and Perfecto, 2011). Furthermore, theories of echolocation call design have indicated that bats foraging in more open space should have difficulty foraging in highly cluttered space (Schnitzler and Kalko, 2001). As we found very few edge and gap bats in primary forest, this raises the possibility that modified habitats constitute not only suitable, but in fact necessary habitats for this guild. The most plausible explanation for the high activity levels in both garden and secondary forest habitats is the high plant diversity and associated high insect abundance (Klein *et al.*, 2002), which is providing favourable foraging opportunities for bats.

Contrary to expectations, we did not find strong evidence that primary forests are essential for narrow-space foragers (Kingston *et al.*, 2003). *H. demissus* and *A. tricuspidatus* were recorded in all disturbed habitats, although they did seem to exhibit a gradated response to changes. Nevertheless, there are several reasons to suspect that primary forests remain an important resource for these bats. Firstly, the importance

of intact primary forest for Hipposideridae species has been found elsewhere in the Paleotropics, where total species numbers was much higher than on Makira (Lane *et al.*, 2006, Furey *et al.*, 2010). Secondly, primary forests on Makira have been shown to be important for bird species (Danielsen *et al.*, 2010). Thirdly, *A. tricuspidatus* has higher activity levels in primary forest than in modified habitats, unlike any other species on Makira. Personal field observations have also noted that another Hipposiderid (*cervinus*) is restricted to primary forest. Finally, we note that the two undetected species known to Makira are Hipposiderids, which are more likely to remain undetected in primary forest. Thus, though our results do not support the importance of primary forests for bats, we refrain from concluding that this habitat is not an important resource for narrow-space bats.

A1.5 Conclusions

We find that modified habitats have conservation value as they provide important foraging resources for echolocating bats; with subsistence gardens being particularly important for edge and gap foraging bats. This finding is in line with the growing body of evidence across taxonomic groups indicating that lightly modified landscapes are worthy of conservation attention (Bhagwat *et al.*, 2008). As found elsewhere in the Paleotropics, our results suggest that monocultures make poor habitats for bats. However, we are unable to make any firm conclusions regarding the importance of primary forest habitat for bat diversity due to the relatively low number of echolocating bat species present on Makira and due to detectability issues in this cluttered environment.

Acoustic methods are a low-effort, low-cost method for monitoring biodiversity, and have often been thought of as a “silver bullet” answering questions of habitat use (Gannon and Sherwin, 2004). In our case, results were often inconclusive, and we

specifically emphasize the problems of detectability which emerged from our results, and this must be addressed if successful evaluations of habitat suitability are to be performed with acoustic methods. However, we note that correcting for detectability is becoming increasingly feasible (see Yates and Muzika, 2006, Gorresen *et al.*, 2008) and estimates of species detectability have recently been published for Neotropical species, enabling further corrections to be made (Meyer *et al.*, 2011). Altogether, this work constitutes the first detailed exploration of anthropogenic effects on mammalian diversity in the Solomon Islands, with findings of important conservation relevance both locally and globally.

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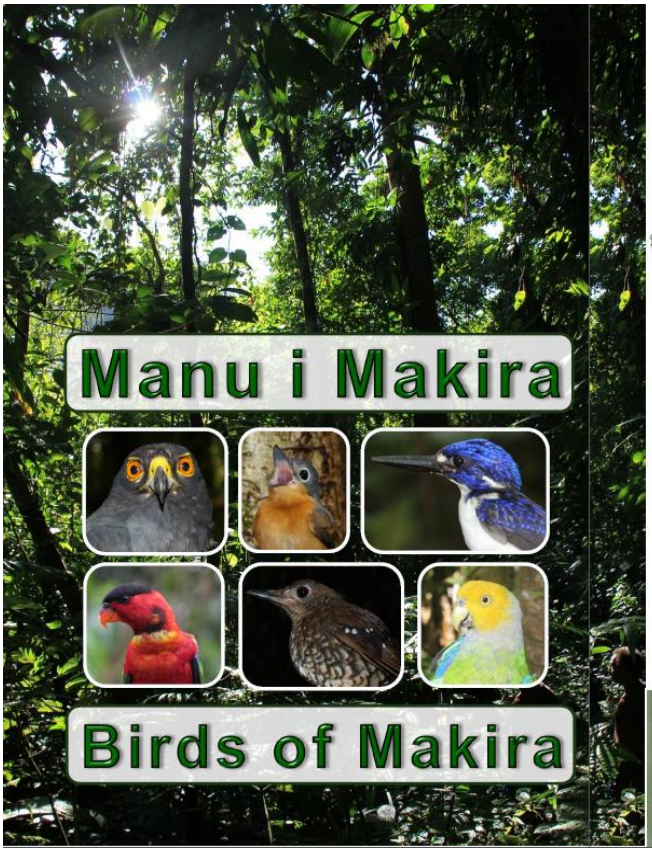
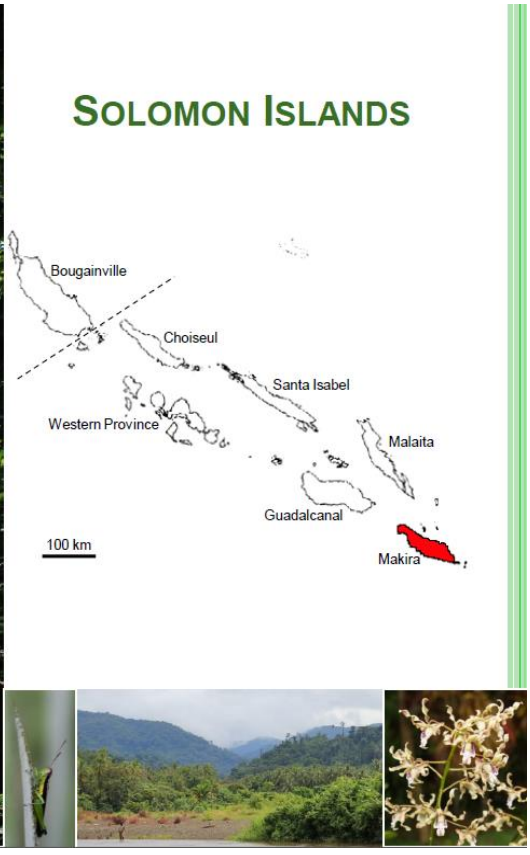
Freda, a research assistant, placing the ultrasonic

monitor (SM2BAT) in the forest for a night of recording

Appendix A2. Extract of Birds of Makira (Manu i Makira)

Available online: *https://static.zsl.org/files/birds-of-makira-web-2095.pdf*





This book is a collaborative effort and is an output from Rufford Small Grants project 11022-1 and Tammy Davies' PhD research on Makira 2012

Produced by Tammy Davies[1,2] **& Dr Rohan Clarke**[3]
[1]St Andrews University; [2]Institute of Zoology; [3]Monash University

Text translated into Kahuan by Br Andrew Wetara

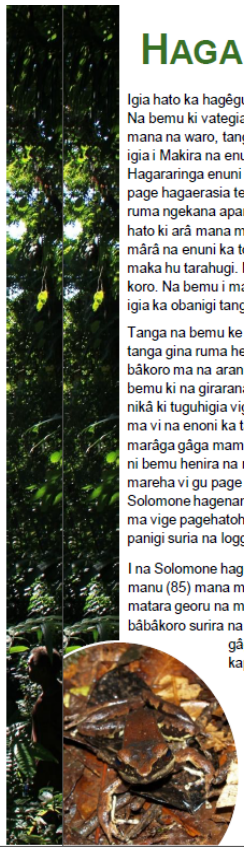
Special thanks to James Taata, Freda Wagoa, Romano Tarohania, Mandy Haga, Nelly Haga & Bassie Wa'apa. And also to Toroa, Huni & Katoro communities, Philip Marupa & the Kahua Association for their cooperation with this research.

Photographs: Tammy Davies, Rohan Clarke (www.wildlifeimages.com.au), Patrick Pikatcha, Chris Filardi, John Ewen, Kevin Parker, Phil Bender, J. del Hoyo/Lynx, Nik Borrow, Paul Noakes, Brian Field, Geoff Jones, Michael Poulsen, Guy Dutson, Ian Reid, Mike Watson, John Carlyon & Birdlife International.

Information on bird size and elevation distribution has been taken from *Dutson (2011) Birds of Melanesia. Christopher Helm, London*

Funded by The Rufford Small Grants Foundation

The Rufford Small Grants Foundation
www.ruffordsmallgrants.org



HAGANEGANA

Igia hato ka hagêguria na bemu tanga na ogaha iaka mana apekoroha iaka. Na bemu ki vategia na mârâ koro i rako tehavê manga na we koro, apanige mana na waro, tangagi na ruma hena ngêra mana meresini ni pasika. Ma igia i Makira na enuni ka rako ka arâ. Tahema na bemu ki mareha ge arari. Hagararinga enuni ga rako ka page obanigi na ngêra nikâ na bemu raha ka page hagaerasia tenigia na mu'u. Nikâ na enuni ga rako ka page obanigi na ruma ngekana apanige i rako ka page tapitapi vato. Ngekana na bemu ki hato ki arâ mana marâ ga mauri ga gâga i rarona ka hato ka arâ nikâ. Na mârâ na enuni ka totora hagarangi igê na matena hato i ngêkê ka arâ hagatê maka hu tarahugi. Ngekana na bemu ki urisi maki vato maki mareha ge koro. Na bemu i mareha ge koro ki bara ma ge oga ke vategia na marâ koro igia ka obanigi tangagi na ôgaha iaka igê na arani tangiana.

Tanga na bemu ke oga ki vategia na ve koro mana a panige, mana varo tanga gina ruma he na ngêra mana meresini ni pasika. Igia ka bâbâsurigi bâkoro ma na arani marâ nge gê. Marupera bâkoro na arani nahuni nga ni bemu ki na girarana (conservation). I Solomone hagenani henia i Makira nikâ ki tuguhigia viga bâbâsurigi na arani marâ hato i bemu. Mareha suria ma vi na enoni ka tako togo tanga gina marâ ni bemu re na ngauha mana marâga gâga mamaru igê, ma suria i Makira ka na hinua iata na arani marâ ni bemu henira na manu. Arani marâ hato nge gê mana manu nikâ ki mareha vi gu page tarahura i tana maveta, tahe ma i katoa i Makira mai Solomone hagenani ki koro getêa. Na arani marâ ni bemu nge gê kasugia ma vige pagehatohato noga, manga na hinua iata ka urisigi se ga aharaga panigi suria na logging vari kierasi se na tauhata na enuni ki erasira.

I na Solomone hagenani ki nahunia na tangahuru gevaru matara kirima na manu (85) mana manu ga goga i Makira ma ka ararina tangahuru gitage matara georu na manu (13). Na girarani na marâ nge gê ki tugu higia viga bâbâkoro surira na arani marâ ni bemu henira na manu ko hanana ga gâga heheurisige ma, manga ga hato i Makira, vikana kapage hato ngavê tenia na maveta.

Na buka ikê ki nahunigi na arani marâ na emoni kabani giraragi, renira na manu mana arani marâ ni bemu, na hinua ita mana mare koro iata i Makira maki manga ihe i gia ka page bâbâsurira bâ koro kohana ga tahitahi tare tanga gia.

INTRODUCTION

We all directly depend on the environment for our livelihoods and health. The environment provides us with many benefits such as clean water, building materials, food and medicines. But the population on Makira is increasing. This means more pressure on the environment. For example, more people will need food so more bush will need to be cleared for gardens. Also, more people will need houses so more trees will be cut down. This leads to the bush slowly disappearing and the animals that live in the bush disappearing too. Resources used to be close to villages, but now people have to walk further to find them. This is an indication that the environment is changing and is becoming degraded or unhealthy. An unhealthy environment is unable to continue to provide the same level of resources that we need to maintain our way of life.

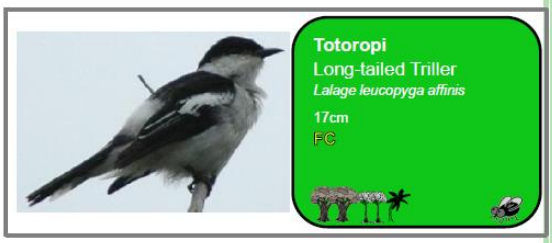
For the environment to continue to be able to provide clean water, building materials, food and medicines, we must look after it and use resources wisely. Protecting the environment and resources is known as *conservation*. The Solomon Islands, and especially Makira, are important areas that need to be conserved, not only because many people depend on their environment for food and shelter, but also because Makira is home to many different types of animals and birds. Many of these animals and birds are found nowhere else in the world and this makes Makira and the Solomon Islands special. An animal that is only found in a certain area, such as an island, is known as *endemic*. Endemic animals can easily become endangered or extinct if their habitat (where they live) changes or is removed, such as by logging or loss of the habitat due to human actions.

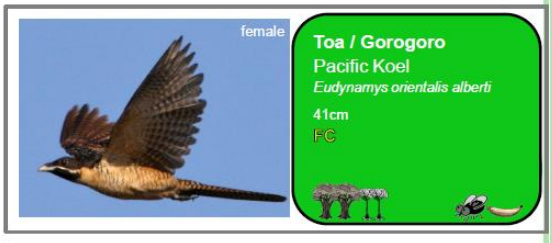
The Solomon Islands contains 85 endemic birds, while Makira is home to 13 endemic birds. This means that it is important to protect the environment to ensure these birds can continue to live here as if they are finished on Makira, they will be finished from the world.

This book contains the common and endemic birds from Makira and we hope you find it useful to learn more about these birds and why Makira is an important place for conservation!

HOW TO USE THIS BOOK

∗ = the photograph was taken at a location other than Makira and the bird shown may differ slightly





Appendix A3. Ethics Approval and Informed Consent

University of St Andrews

University Teaching and Research Ethics Committee
School Of Geography And Geosciences

3 February 2011
Tammy Davies
Geography and Geosciences

Ethics Reference No: *Please quote this ref on all correspondence*	GG7208
Project Title:	Poverty and Ecosystem Services in the Solomon Islands
Researchers Name(s):	Tammy Davies
Supervisor(s):	Ioan Fazey

Thank you for submitting your application which was considered by the Geography and Geosciences School Ethics Committee. The following documents were reviewed:

1. Ethical Application Form — 2 February 2011
2. Letter from Kahua Association — 2 February 2011
3. Verbal Informed Consent Script — 2 February 2011
4. Risk Assessment — 2 February 2011

The University Teaching and Research Ethics Committee (UTREC) approves this study from an ethical point of view. Please note that where approval is given by a School Ethics Committee that committee is part of UTREC and is delegated to act for UTREC.

Approval is given for three years. Projects, which have not commenced within two years of original approval, must be re-submitted to your School Ethics Committee.

You must inform your School Ethics Committee when the research has been completed. If you are unable to complete your research within the 3 three year validation period, you will be required to write to your School Ethics Committee and to UTREC (where approval was given by UTREC) to request an extension or you will need to re-apply.

Any serious adverse events or significant change which occurs in connection with this study and/or which may alter its ethical consideration, must be reported immediately to the School Ethics Committee, and an Ethical Amendment Form submitted where appropriate.

Approval is given on the understanding that the 'Guidelines for Ethical Research Practice' (http://www.st-andrews.ac.uk/media/UTRECguidelines%20Feb%2008.pdf) are adhered to.

Yours sincerely

Dr. Sharon Leahy
Convenor of the School Ethics Committee

UTREC School of Geography and Geosciences Convenor, Irvine Building, North Street, St Andrews, KY16 9AL
Email: ggethics@st-andrews.ac.uk Tel: 01334 463897

Informed Consent

Script prepared to be as simple as possible and for research assistant to read out in Kahuan.

1) I am working with Tammy Davies from St Andrews University to do some research in Kahua

2) This research is looking at the relationship between biodiversity, what people use from the forest and the differences between people such as wealth.

3) This research is important to understand how different people use the forest and how important the forest is to them and how this is changing

4) Your participation is voluntary and you can stop at anytime

5) The information from this study will be summarized in a workshop <location and date>. A report will also be provided to the Kahua Association and Tammy's University.

6) No individual will be identified in any report and all information is confidential.

7) Discuss above

8) Write down name of participants

9) Tammy may also take some photos, which may be used in reports – if you don't want your photo taken then please say.

Appendix A4. Population and social structure of the study villages in Kahua

	Villages		
	Toroa	Huni	Katoro
Total households in village	40	29	15
Households surveyed	32	27	15
Average people per household	5	6	5
Average years of education	6	6.2	4.7
Religion	South Seas Evangelical	Catholic	Catholic
Distance to provincial capital Kirakira (km)	32.4	24.9	21.9
Sanitation	No	No	No
Water supply	Piped water (outdoors, shared)	No piped water	No piped water
Nearest clinic (km)	2.1	5.6	2.6
Nearest primary school (km)	0	0.7	3.9
Nearest secondary school (km)	14.5	7.3	4.3
Boat fare to provincial capital Kirakira (US$)	$14	$8.5	$7

NB: Kirakira is the provincial capital, although relatively small and with limited infrastructure it contains a hospital, numerous stores, airstrip and telecommunication coverage.

Appendix A5. Publication arising from Chapter 2

Co-authour contributions: I was responsible for the research, design and writing of this chapter. All co-authours commented on different versions of this chapter, providing advice on how to improve the content.

Animal Conservation

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REVIEW

Missing the trees for the wood: Why we are failing to see success in pro-poor conservation

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Keywords
biodiversity; poverty; success; interdisciplinary; monitoring; adaptive management.

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Abstract

Pro-poor conservation strategies are touted as a panacea for achieving biodiversity conservation and poverty reduction, yet there remains limited evidence for success in achieving these dual objectives. Is this lack of evidence a sign that the approach is failing or an indicator that we are not capturing appropriate measures of success? In this review, we demonstrate that the lack of evidence for success in pro-poor conservation can be attributed to focusing on the bigger picture and overlooking the details. This has led to two fundamental flaws, (1) an ambiguous use of definitions and (2) inappropriate monitoring, both of which are compounded by limited understanding between traditional scientific disciplines and which render true conclusion of success impossible. Due to the current lack of evidence, pro-poor conservation risks basing decisions on belief rather than evidence, repeating mistakes and missing opportunities to replicate successes. We propose five key recommendations that will facilitate the identification of successful pro-poor conservation interventions and the conditions under which they work. Adoption of these recommendations will ensure pro-poor conservation is built on clear definitions and engages in meaningful participation with rigorous monitoring and reporting of outcomes, aiding replication and scaling up of pro-poor conservation successes in intelligent and evidence-based ways.

Introduction

The current decline in biodiversity is of great concern, not only because of the loss of its intrinsic value and the unknown impacts on ecosystem functioning, but also because of the potential negative repercussions for human well-being (Cardinale *et al.*, 2012; Reich *et al.*, 2012). Addressing biodiversity loss and poverty are international societal and political goals (Sachs *et al.*, 2009; Roe, 2010). Each year the world spends around US$126 billion of official aid tackling global poverty and US$8–12 billion on addressing biodiversity loss (Roe *et al.*, 2011), yet in neither case are these resources considered sufficient to solve these challenges (Roe *et al.*, 2011; Evans *et al.*, 2012). The majority of the world's poor live in rural areas (International Fund for Agricultural Development, 2010) where they depend disproportionately on biodiversity for their survival (Belcher, 2013). This relationship has led to the explicit assumption that conserving biodiversity can help address global poverty, and in light of pressing challenges, such as population growth, overconsumption and climate change, there is a strong need for further integration of the poverty alleviation and biodiversity conservation agendas (Sachs *et al.*, 2009).

The integration of these agendas has so far proved more difficult and more expensive than many had hoped (Adams, 2013). Conservation activities first started to address development issues in the 1970s, motivated by the substantial negative impacts on local people born from earlier 'fortress' conservation activities (Adams *et al.*, 2004). The projects conducted according to this 'people-friendly' approach, broadly termed integrated conservation and development projects (ICDPs) (Blomley *et al.*, 2010), initially attracted substantial support from international development agencies and conservation NGOs, and were rather hurriedly accepted as a panacea for conservation and sustainable development. However, early results proved disappointing, and the approach rapidly fell out of favour (McShane *et al.*, 2011; Miller, Minteer & Malan, 2011). The ICDP label is now less common, but biodiversity conservation and poverty reduction continue to be addressed as dual goals; the extent to which this is so largely depends on the perceived role of poverty in determining the status and threats to the intended conservation target (Sandbrook & Roe,

2013). Adams *et al.* (2004) proposed a typology of four positions conservationists may take to the question of poverty: (1) poverty and conservation are separate policy realms; (2) poverty is a critical constraint on conservation, meaning it must be tackled to achieve conservation goals; (3) conservation activities must not compromise poverty reduction; and (4) poverty reduction is a goal dependent on resource conservation. Recognizing these fundamental differences in value positions can help contextualize the projects rationale, objectives and behaviour of different actors, particularly when they are faced with difficult trade-off decisions between conservation and development goals (Leader-Williams, Adams & Smith, 2011). Truly addressing the dual goals of biodiversity conservation and poverty reduction requires adopting the fourth position detailed by Adams *et al.* (2004), an approach known as 'pro-poor conservation' and defined as 'a people-centred approach that has poverty reduction and livelihood security as core objectives and seeks robust conservation approaches to achieve these' (adapted from: Roe *et al.*, 2003; Roe & Elliott, 2006; Kaimowitz & Sheil, 2007). It builds on the poor's priorities and capabilities, effectively engages all stakeholders in addressing the underlying policy and institutional drivers of environmental degradation and empowers vulnerable groups with the assets, rights and entitlements they need to improve their lives through sound environmental management (Hazlewood, Kulshrestha & McNeill, 2004). Pro-poor conservation can take a number of different forms and encompass a variety of approaches, including: community-based conservation initiatives, direct payments (REDD+) and locally managed protected areas.

Despite biodiversity conservation and poverty reduction being addressed as linked challenges for a number of years, there is growing concern over the lack of empirical evidence for these endeavours (Barrett, Travis & Dasgupta, 2011). An extensive review by Leisher *et al.* (2013*b*), although constrained by the limited number of studies with robust evidence of poverty impacts, was able to highlight ten conservation interventions that had contributed to poverty reduction. However, this review also found that only four of these ten initiatives benefited the poorest (see Table 1 and Leisher *et al.*, 2013*b* for further details). Despite the dearth of empirical evidence for success, there is still broad consensus among conservation professionals that there is a positive link between biodiversity conservation and poverty reduction (Roe *et al.*, 2012). However, without empirical evidence, pro-poor conservation risks basing decisions on belief rather than evidence, repeating mistakes, and missing opportunities to replicate successes (Pullin *et al.*, 2013).

In this review, we synthesize the latest literature from the conservation, development and interdisciplinary realms. Our aim is not to add to the debate on whether conservation can contribute to poverty reduction, which distracts from the real task of improving the effectiveness of conservation–poverty reduction integration (see Roe *et al.*, 2012), but rather to take the issue back to basics and evaluate reasons for the lack of evidence for success. Through this review we hope to clarify key terms and concepts and provide recommendations to move pro-poor conservation forward.

Table 1 Summary of conservation interventions shown to provide poverty reduction benefits and those which benefited the poorest

Intervention	Number of studies	Benefits the poorest
Commercialization of non-timber forest products	> 50	Yes
Community forestry	> 50	Yes
Payments for ecosystem services	10–50	
Nature-based tourism	10–50	
Locally managed marine areas	10–50	Yes
Mangrove conservation	10–50	Yes
Protected-area jobs	< 10	
Agroforestry	10–50	
Grasslands	< 10	
Agro-biodiversity	< 10	

Source: Adapted from Leisher *et al.* (2013b).

Ambiguous use of definitions

Key concepts are often not explicitly defined in pro-poor conservation efforts (Lu, 2010; Roe, 2010; Roe *et al.*, 2013), leading to vague objectives and preventing data aggregation and comparison (Kapos *et al.*, 2008). Poor or narrow definitions are the root cause of limited empirical evidence in pro-poor conservation: how a concept is defined determines what is measured, and what is chosen to be measured determines how success is defined (Leisher *et al.*, 2013*a*).

Poverty

Our understanding of poverty has evolved considerably from its original definitions, which focused on a lack of income or wealth (Sen, 1993; Roe, 2010). It is now viewed as a multidimensional concept, encompassing material deprivation, the lack of access to basic needs such as education, health, nutrition and food security, and the absence of political autonomy and empowerment, as well as the lack of freedom of choice and social inequality (Chambers, 1995; Shackleton *et al.*, 2008; Sunderlin *et al.*, 2008). Interest in multidimensional poverty measurement has been growing steadily over the last decade (Alkire & Foster, 2011; Ravallion, 2011; Ferreira & Lugo, 2012), and mainstream poverty research has become more sophisticated in its handling of this concept, both through a diversification of methods and through more inclusive processes of assessment that include the perspectives of the poor (Addison, Hulme & Kanbur, 2009).

While the set of deprivations to be measured will vary depending on the social context, example indicators can include school attendance, access to safe drinking water, ownership of durable assets and nutrition (Alkire & Foster, 2011). Patterns of associations across multiple dimensions of poverty can be explored through simple tabulations, Venn diagrams or stochastic dominance analysis (Ferreira

& Lugo, 2012); they can also be aggregated into indices, such as the Multidimensional Poverty Index (Alkire & Foster, 2011) or the Human Poverty Index (Dhillon & Kaur, 2010). Poverty can be defined in either relative or absolute terms. Absolute terms measure poverty in relation to the amount of money necessary to meet basic needs, whereas relative poverty is defined in relation to the prevailing standards of living of other members of the society (Scheidel, 2013). The decision whether to use absolute or relative poverty is ultimately a value judgement dependent on the main purpose for which the poverty measure is to be used. Where the relative definition is adopted, it is important to find a locally appropriate understanding of a certain social group or context (Scheidel, 2013). In either case, the chosen dimensions across which poverty has been defined and measured must be explicitly stated, as conceptual and methodological differences in defining poverty can lead to the identification of different individuals and groups as poor (Lu, 2010).

Although reports of pro-poor conservation studies discuss poverty as a multidimensional concept, this is not translated into how poverty is measured, which remains confined to narrow income indicators (Scheidel, 2013; Leisher *et al.*, 2013*b*). This is despite income being known as an insufficient indicator of poverty reduction in isolation and having a low rank among the items by which the poor define their well-being (Chambers, 1995; Brandolini, Magri & Smeeding, 2010; Ferreira & Lugo, 2012). Furthermore, poverty reduction in the income-poverty sense does not capture the complexity of livelihood and well-being-related objectives from local points of view (Berkes, 2013), and these simplistic definitions of poverty have misdirected conservationists regarding the question of community wants and needs, leading to narrowly conceived conservation benefits and incentives for local people (Murphree, 2009; Berkes, 2013).

The dispersion of a distribution of income, consumption or other welfare indicator (i.e. its inequality) is also an important dimension that is largely ignored. Inequality is moving to the forefront of the development policy agenda due to recognition of its relationship to poverty and poverty reduction (Melamed, 2012): inequality directly determines the rate of poverty reduction, and certain types of inequalities (e.g. access to health care, education or markets, or civil and political rights) have a direct causal effect on poverty (Jones, 2009). Equity has largely been addressed indirectly in pro-poor conservation – through implicit assumptions about spreading the costs or benefits, or as a secondary concern, as with *post hoc* comparisons of the equity of outcomes (White *et al.*, 2012). Pro-poor conservation would benefit from a focus on inequality, as this would provide an incentive to focus on those groups that are left behind.

Biodiversity

Biodiversity is defined by the Convention on Biological Diversity as 'the variability among living organisms from all sources including diversity within species, between species and of ecosystems' (CBD, 1992). However, biodiversity is often used to refer to an amount in terms of species and populations or to specific elements of biodiversity rather than variety *per se* (Roe *et al.*, 2013). Biodiversity, like poverty, is a fundamentally multidimensional concept (Lyashevska & Farnsworth, 2012): it can be measured in terms of different components (genetic, population/species, and community/ecosystem; see Table 2). Each of these components has compositional, structural and functional attributes; these categories can be considered the 'three dimensions' of biodiversity (Lyashevska & Farnsworth, 2012). This means that no single level of organization (e.g. gene, population, community) is fundamental, and different levels of resolution are appropriate for different questions (Noss, 1990).

Due to the sheer number of species and the difficulty of sampling many ecosystems, measurements need to be simplified into tractable, quantifiable units that can be compared across time and space (Bradshaw & Brook, 2010). Organism-based metrics that count the number of distinct

Table 2 Indicators of attributes and components of biodiversity with a focus on measures that would be most useful in determining potential effects of human use on biodiversity

Attributes/Components	Composition	Structure	Function
Genetic	Allelic diversity	Heterozygosity Heritability	Gene flow Genetic drift Mutation rate Selection intensity
Population/ species	Species abundance Biomass Density	Population structure, dispersion and range	Fertility Mortality Survivorship Life history Phenology
Community/ ecosystem	Relative abundance of guilds or life forms Proportion of exotic or endemic species	Spatial geometry and arrangement of patch types	Disturbance regimes Nutrient and energy flows Biomass productivity Patch dynamics

Source: Agrawal & Redford (2006).

species in a defined area (species richness) are the most common (Bradshaw & Brook, 2010). However, species richness is arguably a narrow and poor estimate of biodiversity (Lyashevska & Farnsworth, 2012). Metrics that accommodate a broader definition of biodiversity have been developed (summarized in Williams, 2004), and there is a growing recognition of the functional, phylogenetic and taxonomic aspects of biodiversity in conservation biology (Lopez-Osorio & Miranda-Esquivel, 2010; Mouchet *et al.*, 2010; Strecker *et al.*, 2011).

There are two main problems with the use of the term 'biodiversity' in pro-poor conservation. Firstly, where biodiversity has been defined, it is typically measured using a narrow perspective, such as species richness, or a proxy that does not include any ecological information, such as perceptions of change in animal populations or attitudes towards conservation (Agrawal & Redford, 2006). Broader approaches to defining and measuring biodiversity have yet to be incorporated into pro-poor conservation efforts. Conservation measures that aim to enhance a specific attribute or component of biodiversity may have unanticipated effects on other measures of biodiversity; multiple measures targeting specific combinations of attributes and components of biodiversity are therefore needed (Agrawal & Redford, 2006). Secondly, the components used to frame biodiversity are often not clarified, rendering the collected data meaningless. Knowing the biodiversity (however measured) of one place, group or time is not useful in itself; it is the comparable measurements of biodiversity from multiple places, groups or times that can be used to answer crucial questions about how we might best act to conserve it (Purvis & Hector, 2000).

Inappropriate monitoring

Another reason for the lack of evidence for success for pro-poor conservation is the lack of monitoring and inconsistent reporting of outputs or outcomes. Continual and independent evaluation of conservation interventions is a prerequisite to ensuring that conservation is appropriately targeted and effective (Saterson *et al.*, 2004; Sutherland *et al.*, 2004), in addition to ensuring that conservation fulfils its ethical responsibility to do no harm (Barrett *et al.*, 2011). But as with mainstream conservation, pro-poor conservation initiatives have struggled with designing appropriate methods for monitoring and evaluating project progress and outcomes (Blom, Sunderland & Murdiyarso, 2010). This lack of evaluation has restricted opportunities to learn and improve pro-poor conservation through adaptive management (Saterson *et al.*, 2004).

Successful pro-poor conservation strategies occur across a variety of dimensions, including attitudinal, behavioural, economic and ecological (Wamukota, Cinner & McClanahan, 2012). Data that cover only one or two of these dimensions have limited analytical value and can overlook trade-offs between outcomes (Daw *et al.*, 2011; Brooks, Waylen & Mulder, 2012). For example, an extractive reserve may be considered a success by an economist based on increased income for local inhabitants, but a failure by an ecologist or an anthropologist based on critical population decline within the ecosystem and an absence of changed community values regarding conservation (Waylen *et al.*, 2010; Brooks *et al.*, 2012). Consequently, in order to effectively determine success of a pro-poor conservation strategy, measures are needed across the distinct dimensions, as defined by the expected outcomes (Wamukota *et al.*, 2012). Furthermore, monitoring data for pro-poor conservation initiatives is rarely disaggregated for the poorest (or by ethnicity, gender, religion, etc.), yet this would facilitate explicit assessment of how equity influences the ability to achieve project outcomes and produce a more nuanced picture of the intervention impacts on different groups (Halpern *et al.*, 2013). Inappropriate monitoring can be attributed to four key problems: ambiguous definitions, donor pressures, lack of understanding between traditional scientific disciplines and lack of adequate reporting.

Ambiguous definitions

The use of ambiguous definitions cascades to vague objectives and difficulty in developing targets and indicators to gauge performance. Objectives of conservation projects are often not clearly stated or linked directly to individual actions that might be monitored later (Bottrill, Hockings & Possingham, 2011). For example, community-level development activities, such as alternative livelihoods, are often emphasized as an indirect step toward effective long-term biodiversity conservation, but when the link between the activities and the aim are vague, projects tend to focus on ticking off activities, as opposed to monitoring the impacts of these activities (Sayer & Wells, 2004). What, where and how to monitor follows logically from clear and unambiguous objectives and questions (Lindenmayer *et al.*, 2012).

Donor pressures

Effective monitoring is compromised through the influence of donors' demands and priorities (Bottrill *et al.*, 2011) in three main ways. Firstly, there is a mismatch between the short time frame of funding and the long time frame needed to evaluate impacts on socio-economic conditions and the ecosystem, which often take longer than the funding period to undergo detectable change (Barrett *et al.*, 2011; Pullin *et al.*, 2013). Secondly, projects are influenced by donor priorities and emphasize particular kinds of objectives over others, with the objectives that are less attractive to funders left unstated or poorly articulated (Pullin *et al.*, 2013). Pro-poor conservation projects have multiple objectives, and as a result the total monitoring can be a major drain on finite resources, with managers often reluctant to divert scarce resources from action to monitoring (Gardner, 2010). In addition, the inarticulation of all objectives in the funding proposal means there are often inadequate funds available for monitoring the full range of outcomes (Kapos *et al.*, 2008). Thirdly, neither donors nor conservation organizations have created a culture in which monitoring of

outcomes is seen as desirable in its own right (Kapos *et al.*, 2008). Both individual and institutional concerns about exposing shortcomings have served as a strong disincentive for critical evaluation. In some cases there is also an insidious disincentive for claiming or demonstrating success in that perceived improvements may reduce the case for public, political and/or financial support (Pullin *et al.*, 2013). However, Kapos *et al.* (2008) suggest that the capacity for rigorous analysis and synthesis should provide a strong incentive for evaluation to donors and practitioners alike.

Limited understanding between traditional scientific disciplines

Monitoring is further compromised through a lack of understanding of social science research by natural scientists, who often lead pro-poor conservation efforts (Lélé & Norgaard, 2005). The collection of social data, such as people's behaviours or perceptions of change, has followed natural scientists' affinity for quantitative data and large sample sizes that allow statistical analyses and broad generalizations (Drury, Homewood & Randall, 2011). But poor understanding of the aims and scope of qualitative methods can lead to the false assumption that social data can be collected in a straightforward manner and interpreted at face value, leading to misinterpretation of the results (Adams, 2007; Homewood, 2013). Natural science research approaches cannot simply be extended to the social science domain, as paradoxically, an emphasis on quantification of social data may compromise data quality and validity (Adams, 2007; Drury *et al.*, 2011).

Ineffective application of social science research methods has also limited monitoring value due to a lack of meaningful participation. Given the range of different dimensions that need to be considered and in order to ensure data quality, the dimensions on which monitoring should focus should be defined by local people's priorities and local interest in contributing to and making use of the results (Homewood, 2013). Local people's participation and inclusion in pro-poor conservation is rarely meaningful and in extreme cases can verge on coercive (Naughton-Treves, 2012). Therefore, what sets out to be a people-centred approach in reality may only 'involve' local communities as recipients of concessions and development assistance (Lele *et al.*, 2010). This ineffective application of social science research methods is ultimately detrimental to what pro-poor conservation sets out to achieve (McShane & Wells, 2004). Successful initiatives require community objectives to be taken seriously and empowerment of all community members (Murphree, 2009). Better application of social science principles – developing an understanding of local aspirations, refraining from manipulating communities and thinking about trade-offs – would help achieve this (Berkes, 2013).

Lack of reporting

Many project implementers still do not report outputs or outcomes consistently, and consequently there have been few quantitative comparative evaluations (Waylen *et al.*, 2010; Brooks *et al.*, 2012). There is also a potential publication and reporting bias, with unsuccessful cases or metrics less likely to be published (Wamukota *et al.*, 2012). This makes true comparative studies and analyses difficult and strongly limits any attempt to describe relationships between poverty reduction and biodiversity conservation.

Focussing on the details: Seeing the trees in the wood

Success in pro-poor conservation actions cannot be expected while these fundamental flaws persist, and we believe addressing these flaws must now become a priority. We propose five key recommendations to facilitate the collection of an evidence base that will enable the success of pro-poor conservation strategies to be determined (summarized in Table 3).

1. Unequivocal definitions of key terms

Poverty and biodiversity need a multidimensional approach to their definition and measurement, and therefore success also needs to be measured across a variety of dimensions. This will lead to more nuanced questions, such as which groups of the differentiated poor depend on which elements of biodiversity (Roe, 2010; Daw *et al.*, 2011). Poverty, biodiversity and success are context-specific, and the dimensions that have been used to classify them need to be clearly stated, alongside additional contextual conditions that can influence success, such as number of communities, size of population, level of dependence on biodiversity and local governance (Waylen, McGowan & Milner-Gulland, 2009; Wamukota *et al.*, 2012). Baseline data also form part of the context and should be collected at the start of the project to provide a benchmark for comparison with data collected through monitoring of subsequent activities (Bottrill *et al.*,

Table 3 Current problems with evaluating pro-poor conservation and proposed solutions

Problem	Solution
Ambiguous definitions	Unequivocal definitions of key terms (e.g. poverty and biodiversity) (Roe, 2010)
Inappropriate monitoring	Rigorous monitoring that is efficient and appropriate for purpose (Kapos *et al.*, 2008)
	Disaggregated data (Daw *et al.*, 2011)
Donor pressures	Long-term approach for sustainability
	Fundamental shift in donor funding (long-term, flexible, etc.) (Bottrill *et al.*, 2011)
Limited understanding between traditional scientific disciplines	Greater awareness and correct application of social-science principles (Drury *et al.*, 2011)
Lack of reporting/ communication	Systematic reporting of outcomes (whether successful or not) (Hirsch *et al.*, 2011)
	Progress for the poorest groups reported separately (Melamed, 2012)

2011). Further work is required on developing and using standard multidimensional measures of biodiversity as the norm, rather than the exception. Species richness has been shown to be a poor surrogate for biodiversity, and consequently an urgent goal is to catalogue species with their significant functional traits in accessible databases to enable field-collected species lists to serve as a key to estimating biodiversity in its fuller meaning (Lyashevska & Farnsworth, 2012).

2. Rigorous monitoring that is efficient and appropriate for purpose

The idea of rigorous monitoring follows naturally from a thorough understanding of all the dimensions that need to be measured for poverty and biodiversity. A monitoring programme must be designed to be sensitive enough to detect incremental changes and capture the full range of potential outcomes across different dimensions (attitudes, behaviour, ecological and economic; see Margoluis *et al.*, 2009). Appropriate monitoring data will reveal opportunities to learn and develop more responsible management practices (Gardner, 2010). Appropriate monitoring also requires local inputs; equitable participation of all stakeholders is considered central to the adaptive management process and ultimately leads to better results and sustainability (Jacobson *et al.*, 2009).

Monitoring data should be both qualitative and quantitative and collected on different levels and scales (e.g. individual, household and community), as multilevel analyses are critical to understanding the dynamics at the different scales that can affect project outcomes (Brooks *et al.*, 2012). Pro-poor conservation efforts should collect data in as disaggregated a form as possible to facilitate analyses of inequalities and ensure the poorest and most vulnerable are not being left behind. Aggregated data can mask inequalities (UN System Task Team on the Post-2015 UN Development Agenda, 2012), whereas differentiated analyses (e.g. by gender, ethnicity, livelihoods and socioeconomic status; see Daw *et al.*, 2011) can facilitate achievement of conservation outcomes (Halpern *et al.*, 2013) by elucidating the contexts where significant trade-offs are likely to occur (e.g. between the well-being of different people, either between or within communities, or between different outcomes, such as income and food security; Daw *et al.*, 2011).

As project outcomes may not be achieved over the small timescale of the project, indices based on outputs will always be needed (Jones, 2012). Assessing the degree to which intermediate outcomes have been achieved can support adaptive management and provide insights on likely long-term effectiveness of interventions (Kapos *et al.*, 2008; Pullin *et al.*, 2013). For this to be effective, how the delivery of outputs is linked to outcomes needs careful consideration (see planning protocol, Conservation Measures Partnership, 2013). Jones (2012) suggests that for output measures to be more valuable for assessing project success, the linkages between outputs and outcomes, both in project proposals and reports, should be explicitly stated alongside the evidence upon which the assumption is based (Jones, 2012).

3. Long-term approach for sustainability

Pro-poor conservation efforts need to be based on long-term commitment, reflecting the time needed to make the project work (Blom *et al.*, 2010). This will require a level of institutional change among organizations and agencies responsible for funding pro-poor conservation, which will be called upon to (1) provide more long-term and flexible funding, (2) incentivize and promote a culture of adaptive management and prospective ventures in trial and error (Bottrill *et al.*, 2011), and (3) be open to funding 'less attractive' projects that address the needs and threats to the poor and that are not built from predetermined external viewpoints (Kaimowitz & Sheil, 2007).

Poverty reduction can occur through improved governance and strengthening of local institutions, and therefore, for long-term sustainability, pro-poor conservation needs to emphasize community engagement, institution building and the devolution of authority and responsibility to local people (Belcher, 2013). Possibilities and mechanisms for exit (or sustainability) strategy may only become clear after a number of years of successful implementation, although it should be considered in the planning stage, with possible revisions based on achievements during the programme (Young, 2008).

4. Greater awareness and correct application of social science research methods

Greater awareness and correct application of social science research methods could be achieved through collaborations across the social and natural sciences. This would improve pro-poor conservation through (1) the adoption of meaningful participation (for further details see Burns *et al.*, 2004), (2) the facilitation of qualitative approaches that complement quantitative methods through acknowledging the limitations of different research methods in a social and cultural context (Drury *et al.*, 2011), and (3) improved understanding of local-level processes and outcomes, particularly with regard to the complex formal and informal governance of common-pool resources. Use of common-pool resources is structured through institutions ('the sets of formal and informal rules and norms that shape interactions of humans with others and nature'; Agrawal & Gibson, 1999); an analysis of these institutions could offer insights for developing appropriate pro-poor conservation activities and could be achieved through the Institutional Analysis and Development framework (see Imperial & Yandle, 2005; Ostrom, 2007, 2009). This would lead to the recognition of factors that can influence pro-poor conservation outcomes at the local level, including power relations and inherent biases of different actors in the research process (Drury *et al.*, 2011).

Likewise, greater collaboration across the social and natural sciences would benefit poverty-reduction

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Appendix A6. Publication arising from Chapter 4

Co-authour contributions: I was responsible for partly-designing this study, and also for performing the data collection, analysis and the writing of this chapter. IF provided input into the study design. NP & WC provided advice and help when designing the analyses. All co-authours read and commented on all versions of this chapter.

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Who are the poor? Measuring wealth inequality to aid understanding of socioeconomic contexts for conservation: a case-study from the Solomon Islands

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SUMMARY

Understanding the local socioeconomic context is important for the design of appropriate conservation initiatives and associated monitoring strategies, especially in areas with high degrees of inequality, to ensure conservation interventions do not inadvertently further disadvantage vulnerable people. Typical assessments of wealth inequality in remote rural areas are constrained by limited engagement with a cash economy, complex family and tribal ties, and an absence of basic infrastructure. This paper presents a simple participatory approach to measure wealth inequality that does not predefine indicators, such as income or assets, but allows the local people choose the most appropriate indicators. A case study from the Solomon Islands revealed poor households in Kahua were characterized by fewer members, fewer members of working age, and fewer male members than wealthier households. The poor also owned fewer of the locally defined indicators of wealth that were collectively correlated with limited land tenure, and, consequently, conservation or development initiatives that are tied to land in Kahua will be less likely to assist the poorest. Adopting this participatory approach could improve the effectiveness of community-based conservation, through facilitating opportunities to explore local poverty and routes for alleviation.

Keywords: conservation, Melanesia, monitoring, participatory, poverty, Solomon Islands

INTRODUCTION

Conservation interventions aimed at improving the sustainability of natural resource use take place within a complex and dynamic ecological, economic, and social landscape (Dawson *et al.* 2010; Rissman 2011). Understanding these complexities is important for the design of successful conservation interventions, especially in areas with high degrees of inequality, to ensure conservation interventions do not inadvertently further disadvantage vulnerable people (Lawlor *et al.* 2010). In response to the failure of 'fortress' conservation efforts that often had substantial negative impacts on local people, many conservation projects now aim to work with local communities (community-based conservation) and include social objectives, such as poverty reduction, as part of their aims (Hutton *et al.* 2005). However, too frequently community-based conservation initiatives are implemented without fully understanding the local socioeconomic context (Homewood 2013). This ignores the heterogeneity of stakeholders and important factors, such as gender, ethnicity, religion, livelihoods, and reliance on biodiversity, that affect how people are able to respond and interact with conservation initiatives (Agrawal & Redford 2006). Failing to recognize these differences risks unequal distribution of costs and benefits from the intervention, with powerful elites capturing the majority of benefits, and the poor becoming further marginalized (Iversen *et al.* 2006; Saito-Jenson *et al.* 2010). This not only violates the ethical responsibility of conservation to do no harm (Homewood 2013), but is also likely to generate conflict between practitioners and communities, undermine support for conservation and ultimately compromise the long-term success of the intervention (Sommerville *et al.* 2010). Understanding the local socioeconomic context can help mitigate the unequal distribution of costs and benefits from conservation by informing the design of appropriate conservation initiatives and associated monitoring strategies (Barrett *et al.* 2011; Homewood 2013).

Given the unequal, and often highly-skewed distribution of resources and access to their benefits in developing countries, it is evident that researchers must analyse conservation benefits to the poor separately from the rest (or whole) of society (Daw *et al.* 2011), which requires wealth inequality to be measured so the poor can be identified. Thus far, the majority of conservation-based studies looking at poverty have used income as a measure of household poverty (Cavendish 1999; Ambrose-Oji 2003; Fisher 2004; Yemiru *et al.* 2010), mainly because income information is often readily available (Perry 2002). While monetary approaches can be useful, they do not provide the multi-dimensional picture of

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poverty that is necessary to develop targeted conservation and development strategies. Poverty is understood to be a multi-dimensional concept, incorporating elements of political disempowerment, a lack of access to critical investments such as education, and economic exclusion, rather than just low levels of wealth (Sen 1993; Chambers 1995; Gönner *et al.* 2007; McGregor & Sumner 2010; Alkire & Foster 2011). In addition, income data have limitations in both accuracy and measurement, particularly in the context of developing countries where community-conservation projects are based, due to temporal fluctuations in income, inaccuracy in recollection, and sensitivity of certain types of income (such as that derived from illegal extraction). Income may not provide the best indicator of wealth inequality, particularly for short-term studies (see Nielsen *et al.* 2012) often required in community-conservation efforts. Income data also fail to reflect the full amount of resources available to a household, including productive assets (such as livestock) and financial assets (for example savings), which can be used as insurance against income shortfalls (Brandolini *et al.* 2010; Nielsen *et al.* 2012).

Broader definitions and consequently measurements of poverty, such as asset wealth, are widely used in development economics (Carter & May 2001). Filmer and Pritchett (2001) developed an approach to asset wealth measurement in the absence of expenditure data that used an aggregate index based on durable assets owned by households to rank households. Assets provide a better picture of long-term wealth because they accumulate over time, last longer and contribute to the productive capacity of a household through its resource stock (Moser & Felton 2009). Asset-based poverty classifications better predict future income and expenditure than income and consumption measures (Liverpool-Tasie & Winter-Nelson 2011) and are the most important determinant of household choice of livelihood strategy (Ellis & Freeman 2004; Babulo *et al.* 2009; Nielsen *et al.* 2012). In addition, development studies that have examined the empirical relationship between initial inequality and subsequent growth have found land and human capital inequality has a stronger effect than income inequality, suggesting that asset inequality matters more (Birdsall & Londoño 1997; Rodríguez-Pose & Tselios 2010). Asset measures of wealth inequality may thus better inform conservation strategies than income or consumption inequality.

Typically, an asset measures approach uses presence/absence data on ownership of assets that capture living standards (for example radio, television, telephone, bike, motorbike, refrigerator and car ownership; Alkire & Santos 2010) and infrastructure and housing characteristics (such as source of water or sanitation facility; Vyas & Kumaranayake 2006), which may form an index of socioeconomic status (or material style of wealth; Cinner 2009). Measurement is often limited to assets that are in some way measureable, and more intangible assets (such as social capital, access and power) are often ignored. Intangible assets are difficult to quantify because they are linked to the context, and to other complementary assets through which the intangible asset is deployed (Kaplan & Norton 2001; Hulme & McKay 2005). The asset approach usually involves an external assessor determining the kinds of assets to be assessed (Rakodi 2002). This external approach can be less informative for conservation studies, particularly where standard asset lists (for example possession of a radio, TV, fridge or bicycle) are inappropriate (because all households lack basic assets).

Assessments of poverty can either be participatory or non-participatory. Participatory approaches are reflexive, flexible and iterative, and therefore better able than external approaches to facilitate exploring local knowledge and perceptions and encourage learning and empowerment at local levels (Chambers 1992; Cornwall & Jewkes 1995). Participatory approaches to poverty assessments are becoming more widely used in the conservation and development arenas, including methods such as wealth ranking, which involves categorizing households or individuals (Chambers 1994; Laderchi *et al.* 2003). However, people's own assessment of their condition may be biased as a result of limited information and social conditioning (Laderchi *et al.* 2003), or exaggerated in hope of receiving tangible benefits (Krishna 2009). In addition, despite the measures being nominally participatory, the level of participation is usually only extended to a few key stakeholders (McGee & Brock 2001; Naughton-Treves 2012).

Across many remote rural areas there are constraints to the use of typical assessments of wealth inequality, such as limited engagement with a cash economy, strong social networks and complex family and tribal ties, and absence of basic infrastructure and development. To gather appropriate and valid data, an approach that goes beyond monetary, presence/absence of standard assets, and key informant approaches is required. With this paper, we aim to present a simple participatory approach to measure wealth inequality that does not pre-define the indicators to be used, but enables local people to identity them. This ensures a flexible and inclusive method, providing a perspective on poverty that is sensitive to local contexts, while simultaneously remaining straightforward and replicable for remote rural community-conservation projects. We demonstrate our approach using a remote and data-deficient region of the Solomon Islands, where there was no prior information on poverty. Our specific objectives were to: (1) identify locally appropriate indicators of wealth, (2) assess whether these indicators are able to represent variation in wealth within communities, (3) compare the indicators with annual monetary income and expenditure, and (4) determine the key predictors of poverty.

METHODS

Study area

The Solomon Islands are an archipelago in the South-West Pacific (Fig. 1) that contains one of the last remaining tracts of coastal tropical rainforest (Bayliss-Smith *et al.* 2003) and is part of the East Melanesian Islands biodiversity hotspot

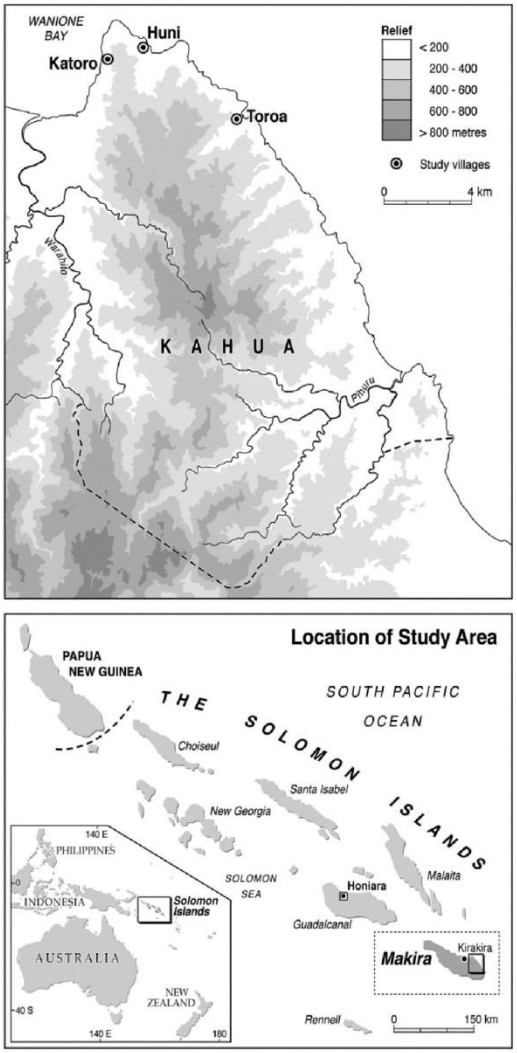

Figure 1 Location of Kahua region of Makira, Solomon Islands.

(Myers *et al.* 2000). These islands are undergoing rapid environmental and social change, with an economy heavily based on extractive industries, coupled with one of the highest population growth rates in the world (2.6% per annum; UNICEF [United Nations Children's Fund] 2011).

This study was conducted in the Kahua region (Wards 12 and 13; 162°0–162°15 E, 10°25–10°40 S) of Makira Island (formally San Cristobal). Makira is the fourth largest island of the archipelago, with an area of 3191 km^2, and consists of a narrow coastal plain with steep forested central ridges with altitudes up to 1200 m (Allen *et al.* 2006). There is limited infrastructure, with no roads and only a limited number of high-frequency radios for communication. Transport to the provincial capital of Kirakira (access to main market and hospital) is either by foot or fibreglass boats with small outboard motors. Lack of market access is a major constraint on economic development (Allen *et al.* 2006).

The Kahua region has approximately 4500 inhabitants across 42 communities, mostly located on the coast. The main livelihood strategy is subsistence agricultural production, supplemented by fishing and exploitation of a wide range of species. Kahua is experiencing rapid social and environmental change through an increasing population, desire for monetary prosperity, a loss of social cohesion (Fazey *et al.* 2011) and a loss of traditional methods of natural resource use and management (Ministry of Environment Conservation and Meteorology 2008). Changes in primary productivity suggest significant ecological change at a landscape scale (Garonna *et al.* 2009) and at the local level, the availability of forest and marine resources may be declining with reports of falling crop yields and increasing incidence of pests and diseases (Bourke *et al.* 2006; Fazey *et al.* 2011).

The market economy was introduced to the Solomon Islands far later than in other developing countries (Furusawa & Ohtsuka 2006). Interaction with the cash economy in Kahua is limited, but increasing, mainly through the payment of school fees, transport and imported foods. Households engage in a range of income-generating activities, including the sale of agricultural produce, handicrafts and livestock (chickens and pigs), and the production of cash crops (copra and cocoa). Participation in these income-generating activities requires access to land, which is predominantly customary-owned across Melanesia, with tenure established through genealogy (Fazey *et al.* 2011). The Solomon Islands have a traditionally male dominated society, and men continue to dominate all sectors of society from political posts to village chiefs (Fazey *et al.* 2011; Mataki 2011) and consequently there are major gender inequalities. Men also dominate most income-generating activities and tend to have a lower commitment (than women) to spending on the health and education of their children (Gibson 2000; McMurray *et al.* 2008; Macintyre 2009). Cultural traditions remain strong, and as with other Pacific Island economies there is an emphasis on redistributive activities, with most households involved in tribe-specific networks that give and reciprocate goods and services, termed the 'wantok system' (Gibson 2006).

Data collection

Our research approach was broadly exploratory and inductive, with a combination of focus groups, discussions and a widely-scaled household survey. It aimed to facilitate exploration of local knowledge and perceptions using deliberative methods that in Kahua are more closely aligned to social deliberative ways in which people communicate (Fazey *et al.* 2010; Kenter *et al.* 2011). Data were mostly collected by five local villagers, trained as facilitators and closely supervised by T. Davies;

these local facilitators were essential for maintaining trust between researchers and communities, engaging with local communities, and translating information. Data collection methods were simplified to account for the facilitators' low levels of education and limited ability to simultaneously translate and record information, while also designed to capitalize on their local cultural and social expertise and knowledge (Fazey *et al.* 2011).

Focus groups were conducted in six villages, with one coastal and one inland village sampled from each of the eastern, central and western localities. All village inhabitants were invited to participate. A total of 12 focus groups were conducted, which included a total of 109 participants, with an average of nine per group. Focus groups lasted on average *c.* 3 hours; all were conducted in local language (Kahuan), and separately for men and women, to manage gender-based power relationships. Due to low levels of literacy, informed consent was sought verbally from all participants at the start of the focus group. After an ice-breaker exercise, participants were asked to identify different wealth groups within their community. All groups identified three different categories: poor, average and wealthy. In groups, participants were then asked to identify items or characteristics that changed across these categories (indicators of wealth). Groups were asked to choose an item close to hand, such as leaves, to represent their chosen indicators, which were then brought together for discussion. How these indicators changed across the wealth categories was then discussed. The total list of indicators from all focus groups was presented and discussed at a workshop, with 30 participants from across Kahua, where in groups of three, participants were given five stones and asked to rank the indicators they considered the best. This led to a shortlist of five asset indicators.

A household survey was used to collect information on ownership of the top five asset indicators at the household level. A household was defined as people living together and sharing meals. The household survey was piloted in April 2011, refined and then conducted across 74 households from three communities in February 2012 and July 2012. All households were surveyed in each of the three communities. The head of the household was interviewed, or if unavailable another adult from the household was interviewed. Basic information on household social structure including composition and levels of education was collected, and in January and July, households were also asked to recall major sources of income and expenditure in the last six months. Income and expenditure data were then combined for a crude annual figure. Where there were differences in the information collected (for example household members or age) between January and July, the average value was used for analysis.

Analyses

All data were analysed with R v2.15.1 (R Core Team 2013). Local indicators of wealth were identified as household ownership of number of pigs, chickens, coconut trees, cocoa trees and gardens. A principal component analysis (PCA) was applied to household data on ownership of these locally-defined indicators of wealth; the indicators are all continuous variables well suited to PCA. The factor scores from the first principal component (the vector that provides the most information about the variables) were used as the socioeconomic status index (wealth score) for each household. The higher the wealth score, the higher the implied wealth of the household. Differences in wealth score between villages were determined using an ANOVA.

To explore the variation in household demographics, a PCA was also applied to the household social structure data (number of household members, number of household members < 18 years, age of household head, education of household head and dependency ratio (number of dependents [0–14 or > 65 years] to the working-age population [15–64 years old]) as a first step to determine the factors explaining most of the variation within the data. We did not consider the first axis of the household social structure PCA as a factor explaining the variation of the first axis of the household asset PCA, as we aimed to assess how each component of the household asset dataset related to wealth inequality between households. To identify the main characteristics of the poor, the constructed household wealth score was then included as a continuous independent variable in a general linear model to explore the relationship between the wealth score and the household social structure: number of household members, age of household head, education of household head, gender of household head, proportion of males in the household and the dependency ratio. All possible combinations of main effects, followed by combinations of interactions were explored and then compared using Akaike's information criterion (AIC) values, which were compared among all possible combinations of explanatory variables. AIC is an evidence factor that is corrected for model complexity. Weighting AICs can be used to assess the model that best fits the data by approximating Kullback-Leiber information loss to see how changing the model affects the fit (Bradshaw & Brook 2010), with a small value representing a better fit of the model to the data. To avoid model selection uncertainty where there were rival models, weighted averages of parameter estimates were calculated following Burnham and Anderson (2002). General linear models were used to compare the wealth score with income and expenditure, with the strength of the correlation assessed using Spearman rank correlation and R-squared values.

RESULTS

Data was collected from 74 households across three communities (Table 1). Respondents had a mean age of 47.5 ($\pm$ 15.0) years, with an average of 5.6 ($\pm$2.4) years of education. Households had an average of 5 members ($\pm$ 2.1), with a mean of 2.4 ($\pm$1.6) children (those under 18 years).

Focus group discussions indicated that wealthier households owned more of the locally-defined indicators, which was corroborated with analysis of asset ownership

Table 1 Population and social structure of the study villages in Kahua.

Factor	*Villages*		
	Toroa	*Huni*	*Katoro*
Households surveyed (n)	32	27	15
Average people per household	5 ± 1.5	6 ± 2.6	5 ± 2.1
Average years of education	6 ± 1.4	6.2 ± 1.9	4.7 ± 1.9
Religion	South Seas Evangelical	Catholic	Catholic
Distance to Kirakira (km)	32.4	24.9	21.9
Sanitation	No	No	No
Water supply	Piped water (outdoors, shared)	No piped water	No piped water
Nearest clinic (km)	2.1	5.6	2.6
Nearest primary school (km)	0	0.7	3.9
Nearest secondary school (km)	14.5	7.3	4.3
Boat fare to Kirakira (US$)	14.0	8.5	7.0

Table 2 Descriptive information gathered from all focus groups on how each indicator changes across the wealth categories (only main indicators identified at the workshop).

Indicator	*Poor*	*Average*	*Wealthy*
Pigs	No space for pig; lazy	Feed pigs coconuts; not always enough to feed them; no fence	More than five pigs; fence for pig; enough food to feed pig; sells for money
Chickens	No chickens; lazy	Some chickens	Lots of chickens; chicken coup
Coconut	No trees; asks for coconut; steals coconuts	Some trees	Lots of trees; always cooks with coconut milk
Cocoa	No trees	Some trees; 1–2 areas	Always sells to the ship
Gardens	Lazy; depend on others; don't plant much (cassava and banana)	2–3 gardens; plant 3+ crops; different crops in different gardens	5–10 gardens; doesn't use same garden each season; variety of foods

Table 3 Wealth factor scores from the principal components analysis of locally-identified indicators of poverty.

Variable	*Wealth factors*		
	1	*2*	*3*
Number of pigs	0.49	− 0.07	− 0.03
Number of chickens	0.53	0.13	− 0.08
Number of coconut trees	0.45	0.37	− 0.41
Number of cocoa trees	0.46	0.03	0.42
Number of gardens	0.24	− 0.65	0.44
Average size of gardens	0.09	− 0.65	− 0.68

Table 4 Wealth factor scores from the principal components analysis of household social structure.

Variable	*Wealth factors*		
	1	*2*	*3*
Household number	− 0.56	− 0.09	− 0.05
Number under 18 years	− 0.62	− 0.02	0.06
Proportion of Males	− 0.16	0.19	− 0.94
Age of head of household	0.08	− 0.67	− 0.25
Education of household head	− 0.14	0.68	0.08
Age dependency ratio	− 0.51	− 0.22	0.21

(Table 2). PCA of these assets generated three components that together explained 71.4% of the variation (Table 3). The first component was composed of chickens with the greatest positive loading, followed by number of pigs, number of cocoa trees and number of coconut trees; these factors explained 36% of the variation in the data. The second component, with positive loading from number of coconut trees and strong negative weighting of garden number and garden size explained 20% of the variation, indicating less variation in gardens across the different wealth categories. The third component, explaining 15% of the variation, had a positive loading from number of gardens and number of cocoa trees, and a high negative loading from garden size and number of coconut trees.

Based on the factor scores from the first principal component, wealth scores for households ranged from –2.07 (poorest) to 5.40 (wealthiest) (mean = 0.00 ± 1.5). Villages did not differ in wealth scores (ANOVA $F = 1.4$, $df = 2$, $p = 0.25$) and therefore all analyses refer to grouped data.

A PCA of household social structure data generated three components that together explained 79 % of the variation (Table 4). The first component consisted of negative loading from number of household members, number under 18 years and the dependency ratio; the first component of this PCA

Table 5 Composite model of the strongest predictors of household wealth scores.

Parameter	*Estimate*	*SE*	*90% CI*	
			Upper	*Lower*
Intercept	− 1.72	0.39	− 1.09	− 2.36
Household number	0.37	0.05	0.46	0.29
Age dependency ratio	− 0.004	0.001	− 0.002	− 0.01
Proportion of males	0.02	0.002	0.02	0.01

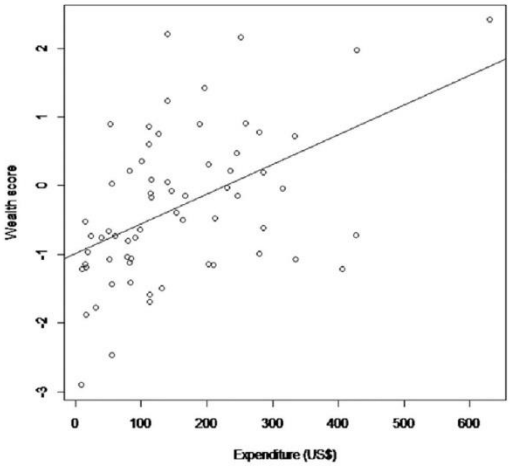

Figure 2 Wealth score versus annual monetary expenditure.

explained 41% of the variation in the data. The second component had a positive loading from education of household head and negative loading from age of household head; the second component of this PCA explained 22% of the variation in the data. The third component had a strong negative loading from the proportion of males in the household; the third component of this PCA explained 16% of the variation in the data.

AIC model weights revealed the household social structure data, modelled as main effects, which best explained the variation in wealth scores were number of household members, age dependency ratio and proportion of males. A higher number of household members, lower age dependency ratio and higher proportion of males were associated with a higher wealth score. Based on Akaike weights, there was a rival model composed of number of household members and age dependency ratio. To avoid model selection uncertainty, weighted averages of parameter estimates were calculated (Table 5).

There were positive correlations between wealth scores and monetary income ($p = 0.006$, $R^2 = 0.11$), the strongest being between wealth scores and monetary expenditure ($p < 0.0001$, $R^2 = 0.24$; Fig. 2).

DISCUSSION

Our participatory asset measurement method avoided typical constraints to assessments of wealth inequality in remote rural areas, such as limited interaction with the cash economy, in addition to avoiding biases associated with external approaches. Our approach provided key insights into characteristics of poor households where there was no prior information on poverty in a culturally sensitive manner that enabled participants to express their views on which indicators were important. Household asset wealth was particularly well correlated with household expenditure, which tends to be a better metric than income because households can smooth their expenditure during a temporary low-income period by borrowing or using savings (Perry 2002). As we only collected a crude measure of household expenditure, more detailed data would be expected to improve the strength of this correlation. However, our participatory approach to asset measurement goes beyond monetary metrics by providing better characterizations of the poor, which in turn provides further insights for the design and implementation of appropriate conservation projects and poverty reduction policies.

The poor in Kahua owned less of the locally-defined indicators of wealth, particularly chickens, pigs, coconuts and cocoa trees. Little is known about rural poverty in Melanesia; however these assets reflect traditional Melanesian symbols of power. For example, pig ownership and pig killing traditionally conveyed status, wealth and informal power in Melanesia (Miles 1997) and pigs are still culturally important in the region, remaining currency for major transactions (Glasse 1959; Miles 1998), including compensation payments and bride price that are commonly applied across the Solomon Islands. Food produce has long been used as a display of power, prestige and competition in Melanesia (Roscoe 2000), with the group with the largest number and size of pigs, food crops and cooked food commanding the most respect (Nanau 2011).

Analysis of ownership of these assets also provided insights into the household characteristics of the poor, whose households had fewer members, a higher age dependency ratio and a lower proportion of males. In fact the poorest households contained no male members (older female living with young female child); other studies have also found female-headed households to be over-represented among the poor (see Buvinić & Gupta 1997; Biewen 2006; Medeiros & Costa 2006). Our participatory research approach enabled additional information to be elicited that would have been difficult to achieve otherwise. Focus group discussions revealed that people felt the poor's social position could be improved through hard work, and a recurring theme was that the poor were lazy. For example, they might have access to land, but did not necessarily put in the effort to cultivate it, and therefore depended on exploiting the wantok system. Views that the poor are lazy are common (see for example Lockwood 2002). However, although the poor may appear lazy, they may in

fact be marginalized in some way, which means that they are unable to capitalize upon opportunities. For example they may have low personal empowerment (for example low confidence or social skills), or may not conform to social norms or abide by the same values as the rest of society (Applebaum 2003). In addition, the poor may not have access to land, for example if they are immigrants from other areas or families of men who have married into the region.

These results may help planning of appropriate community-based conservation and development initiatives to benefit the poorest. The locally-defined indicators of wealth are collectively correlated with land tenure. Thus, a higher wealth score can be seen to equate to ownership of, or access to more land, and consequently conservation and development initiatives that are linked to land will naturally favour uptake by the wealthy, whereas the poor may be unable to invest or allocate land for such schemes (Corbera *et al.* 2007; Börner *et al.* 2010). Yet, current development activities in Kahua are focused on the promotion of cash crops, activities which are unlikely to benefit poor households that have less land and are thus less likely to directly participate in these initiatives. In addition, monetization of resources can increase gender inequalities, adversely impacting women, which is of concern for poverty alleviation efforts in Kahua, where poor female-headed households could become further marginalized. The commodification of natural resources (for example through the introduction of cash crops) has shifted the Melanesian relationship with land from cultural to economic, and this shift is eroding social cohesion, with property rights currently a major source of conflict across Melanesia (Bonnemaison 1984; Foale & Manele 2004; Fazey *et al.* 2011). Given the assets and characteristics of poor households in Kahua, cash payments for conservation (such as payments for ecosystem services) are unlikely to be an appropriate conservation strategy there, because they could increase community conflicts, ultimately undermining conservation activities. Strategies that focus on small-scale resource management, balancing food security and conservation, such as agroforestry and locally-managed marine areas are likely to be more appropriate for the social context in Kahua. Understanding the local socioeconomic context could help develop an appropriate enabling environment with interventions to improve people's capabilities and conditions, such as empowerment programmes and land reform (see McGregor & Sumner 2010).

Community conservation projects are often constrained by time and resources, with a limited portion of these available for monitoring activities (Gardner 2010). Our asset-based approach within a participatory framework is well suited approach to community-based conservation projects in areas with low levels of literacy and resources because it can collect valid and reliable data in an easily replicable manner. The participatory approach also provides an excellent starting point for discussing inequalities, and providing insights into how they can be alleviated or managed (Moser & Felton 2009). Findings from this approach can then be used to assist decision making on how best to target the poor and also as an input to other research problems, such as the relationship between wealth and observed behaviours, for example use of destructive fishing gear (Cinner 2009), uptake of conservation initiatives (Brandolini *et al.* 2010) and livelihoods (Reardon & Vosti 1995). For those community-based conservation projects that also aim to reduce poverty, longitudinal asset data can be used to monitor and determine transitions out of poverty. Although we used asset measures to provide an initial assessment of wealth inequality, this approach can also be employed in community-conservation projects before and after an intervention as part of monitoring activities to record longitudinal asset data or 'asset dynamics', which can help elucidate transitions out of poverty (Carter & Barrett 2006; Adjei *et al.* 2009). An approach for assessing household strategies for poverty alleviation has been pioneered by Krishna (Krishna & Shrader 1999; Krishna 2009).

Although our approach goes further than basic income measures of wealth inequality commonly used in community-conservation projects, it is still restricted to material dimensions of poverty. Our approach was unable to distinguish between important capabilities, for example those who do not have access to land, and those who have access to land but choose not to cultivate it. Less tangible dimensions of poverty, such as social capital and power, were also not reflected in our assessment. Our approach is not a panacea and further research is required into advancing poverty measurement that is better able to capture both tangible and intangible aspects of deprivations. However, it did provide insights into how poverty is viewed in the region, which appears to be based heavily on traditional hierarchies and symbols of power (pigs), and therefore the locally-identified wealth assets may also be a proxy for power; although the extent to which these assets reflect power should be further explored. Social capital is the most commonly cited intangible asset (Moser & Felton 2009), yet kin and friendship networks are often the most important relationships that households mobilize to reduce vulnerability (Bacon 2005). The wantok system is an important informal institution in Melanesia for social cohesion and its contribution to balancing wealth inequality, and its ability to function as a support network, should be further explored using approaches that go beyond asset measures (Krishna & Shrader 1999).

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Appendix A7. Household survey form used for data collection

NB: In the field, this was printed landscape and included more rows and space for notes.

Name	Age	Occupation	Years at School	School level

In the last 2 weeks have you or your family used any items from the bush for MEDICINE?

Y / N

What	Amount	For what sicki	Collected/Purchased	Cost

In the last 2 weeks have you or your family used any items from the bush for BUILDING MATERIALS? Y / N

What	Amount	Collected/Purchased	Cost

In the last 2 weeks have you or your family taken anything from the BUSH or SEA or RIVER for FOOD? Y / N

What	Amount	Collected (where)/Purchased	Cost

All food eaten YESTERDAY in your household: (include snacks, fruits nuts etc)

	Food	Amount (use market measure: pile, bundle, number etc)	Number of people eating	Where food from? (store/garden/sea/bush/gift etc)	Cost
What now 1st kaikai you kaikaim?					
After that?					
After that?					

Now I'd like to ask you about anything RECEIVED or GOT by anyone in your household

YESTERDAY:

Include: food, drink, medicine, kaliko, school materials.	Item (what?)	Paid or Gift	How many	Cost ($)

Please could you tell me any things your household SOLD YESTERDAY? Include Copra, Cocoa, food, items from the bush, things you have made (such as basket, sleeping mat, axe handle etc.)

What	How obtained? (garden, bush, sea, handmade)	Amount (e.g. number, KG)	Cost ($)	Who bought it?

Any notes:

Appendix A8. Total species list for all wild foods consumed recorded from 24hr and two-week recall data. Species names have been included where these were known and confirmed (as local names were often used to refer to more than one species). N.B. Species only recorded from two-week recall data are marked *

	Species			
	Common name	Local name	Latin name	
Terrestrial plants	Sandpaper cabbage	Avusi	*Ficus copiosa*	
	Tree fern	Boroto	*Cyathea sp.*	
	Tahitian chestnut	Mabe	*Inocarpus fagifer*	
	Breadfruit	Parâ	*Artocarpus altilis*	
	Coconut	Ni	*Cocos nucifera*	
	Geke	Geke	*Polyscias fruticosa*	
	Vegetable fern	Gogona	*Diplazium esculentum*	
	Fungi	Karinga	*Volvariella sp.*	
	Spinach jointfir	Suga	*Gnetum gnemon*	
	Ngali nut	Angari	*Canarium indicum*	
	Cutnut	Hara	*Barringtonia edulis*	
	Starfruit	Êri	*Averrhoa carambola*	
	Pineapple	Penawo	*Ananas comosus*	
	Papaya	Memeapu	*Carica papaya*	
	Guava	Guava	*Psidium guajava*	
	Horseradish leaf	Bonio	*Moringa oleifera*	
	Mango	Gai	*Mangifera mangus*	
	Ofenga	Gohere	*Pseuderanthemum whartonianum*	
	Climbing swamp fern	Ano	*Stenochlaen palustris*	
	Malay apple	Gahiga	*Eugenia malacencsis*	
	Vine seed	Hinemora	*Intsia bijuga*	*
Terrestrial animals	Northern cuscus	Huto	*Phalanger orientalis*	
	Pacific flying fox	Roke ni ware	*Pteropus tonganus*	*
	Makira flying fox	Aohinua	*Pteropus cognatus*	
	Bronze Ground Dove	Apungake	*Gallicolumba beccarii*	*
	Night Heron	Ko	*Nycticorax caledonicus*	*
	Prehensile-tailed skink	Gunu	*Corucia zebrata*	
	Red-knobbed Imperial pigeon	Barisumata	*Ducula rubricera*	
	Melanesian Megapode	Auwê	*Megapodius eremita*	*
	Coconut crab	Kasusu	*Birgus latro*	
Marine	Hawksbill turtle	Garohe	*Eretmochelys imbricate*	
	Green turtle	Garohe	*Chelonia mydas*	*
	Turtle	Garohe	unknown sp.	*

	Tropical abalone	Gano	*Haliotis* sp.	
	Chiton	Huruga	unknown sp.	
	Green turbo shell	Ariri	*Turbo* sp.	
	Skipjack tuna	Bonito	*Katsuwonus pelamis*	
	Fish	Manula; Ehuhu	unknown sp.	
Riverine	Crayfish	Maora	unknown sp.	
	Eel	Awowo	*Anguilla* sp.	*
	Crustacean	Mera; Harua	unknown sp.	*
	Fish		unknown sp.	*

Appendix A9. Photographs of the five different land uses surveyed in Chapter 6:

1: Intact Forest, 2: Secondary Forest, 3: Garden, 4: Mixed-cocoa, 5: Monoculture cocoa

Appendix A10. Species guilds for all bird species recorded during surveys on Makira, Solomon Islands * Where relevant trinomials are provided.

Common name	Scientific name	Endemic	Micro-habitat	Prey	IUCN
Barred Cuckooshrike	*Coracina lineata makirae*	Makira	C	O	
Beach Kingfisher	*Todiramphus saurophagus admiralitatis*		AQ	V	
Brahminy Kite	*Haliastur indus flavirostris*	Mel.	AE	V	LC
Bronze Ground Dove	*Gallicolumba beccarii solomonensis*	Mel.	T	FR	LC
Buff-banded Rail	*Gallirallus philippensis meyeri*	Mel.	T	O	
Cardinal Lory	*Chalcopsitta cardinalis*	Mel.	C	FR	LC
Cardinal Myzomela	*Myzomela cardinalis pulcherrima*		C	N	LC
Chestnut-bellied Imperial Pigeon	*Ducula brenchleyi*	Mel.	C	FR	VU
Chestnut-bellied Monarch	*Monarcha castaneiventris megarhynchus*	Mel.	CU	I	LC
Coconut Lorikeet	*Trichoglossus haematodus massena*		C	FR	
Collared Kingfisher	*Todiramphus chloris*		C	V	LC
Crested Cuckoo-dove	*Reinwardtoena crassirostris*	Mel.	C	FR	NT
Dollarbird	*Eurystomus orientalis crassirostris*		C	I	LC
Eclectus Parrot	*Eclectus roratus*		C	FR	LC
Glossy Swiftlet	*Collocalia esculenta*		AE	I	
Island Imperial Pigeon	*Ducula pistrinaria pistrinaria*		C	FR	LC
Long-tailed Triller	*Lalage leucopyga affinis*	Mel.	C	I	
Mackinlay's Cuckoo-Dove	*Macropygia mackinlayi mackinlayi*	Mel.	C	FR	LC
Makira Cicadabird	*Coracina salomonis*	Makira	C	O	
Makira Honeyeater	*Meliarchus sclateri*	Makira	C	N	
Makira Starling	*Aplonis dichroa*	Makira	C	FR	
Melanesian Megapode	*Megapodius eremita*	Mel.	T	O	LC
Metallic Starling	*Aplonis metallica nitida*		C	FR	
Mottled Flowerpecker	*Dicaeum tristrami*	Makira	CU	N	LC
Nankeen Night Heron	*Nycticorax caledonicus mandibularis*		AQ	FI	LC
Oriole Whistler	*Pachycephala orioloides christophori*	Mel.	C	O	
Pacific Baza	*Aviceda subcristata bismarckii*		C	V	LC
Pacific Koel	*Eudynamys orientalis alberti*		C	O	LC
Pacific Swallow	*Hirundo tahitica subfusca*		AE	I	

Pale-vented bush-hen	*Amaurornis moluccana ultima*		T	O	
Pied Goshawk	*Accipiter albogularis albogularis*	Makira	C	V	
Red-knobbed Imperial Pigeon	*Ducula rubricera rubricera*	Mel.	C	FR	NT
Rufous Fantail	*Rhipidura rufifrons russata*	Makira	CU	I	
Singing Starling	*Aplonis cantoroides*		C	FR	
Solomons Sea Eagle	*Haliaeetus sanfordi*	Mel.	AE	V	VU
Sooty Myzomela	*Myzomela tristrami*	Makira	C	N	LC
Stephan's Emerald Dove	*Chalcophaps stephani stephani*		T	FR	LC
Uniform Swiftlet	*Aerodramus vanikorensis vanikorensis*		AE	I	
Variable Dwarf Kingfisher	*Ceyx lepidus gentianus*	Makira	U	V	LC
White-collared Monarch	*Symposiachrus vidua vidua*	Makira	CU	I	
White-headed Fruit Dove	*Ptilinopus eugeniae*	Makira	C	FR	NT
White-rumped Swiftlet	*Aerodramus spodiopygius*		AE	I	
Willie Wagtail	*Rhipidura leucophrys*		T	I	LC
Yellow-bibbed Fruit Dove	*Ptilinopus solomonensis solomonensis*	Makira	C	FR	LC
Yellow-bibbed Lory	*Lorius chlorocercus*	Mel.	C	FR	LC

Endemic: Mel.= Melanesia

Microhabitat: AQ = water and water edge; AE = aerial; C = canopy;

CU = canopy/understory; U = understory; T = terrestrial.

Prey: FR = fruit and other vegetative material; N = nectarivore; O = Omnivore;

I = insects; V = vertebrates.

IUCN RedList status: VU = vulnerable; NT = near threatened; LC = least concern.

* Information collated from Dutson (2012), Kratter *et al*. (2001), IUCN (2013) and our own extensive field experience.

Appendix A11. Semi-structured interview questionnaire

Have you always lived in this village?

If no, where did you move from? What year did you move?

Have you noticed any changes in the bush since you were a child?

(pointers if needed: trees, animals, birds, wild foods)

If yes, can you remember when you noticed a big change?

(pointers if needed: WWII (1945), Independence (1978), Cyclone Namu (1986))

Have you noticed any changes in the river and/or sea since you were a child?

Have you noticed any changes within communities in Kahua since you were a child?

(pointers if needed: size, kastom, men, women, youth)

If yes, can you remember when you noticed a big change?

Have you noticed any changes in gardens since you were a child?

(pointers if needed: crops, distance, size, disease)

Have you ever been hunting? If yes, what did you hunt?

Have there been any changes in hunting, since you were a child?

(pointers if needed: methods, numbers, species)